Environmental Economics and Decision Analysis: An Overview of Recent Results

Mercedes Bertomeu e Carlos Romero*

Abstract

This paper aims to show the extensive applicability of decision analysis tools in environmental economics. A series of environmental situations, where the use of methods based on decision analysis have shown their superiority with respect to those based on cost-benefit analysis are presented throughout the paper. In this sense, the following situations will be commented: a) the appraisal of environmental assets, b) the optimal provision of environmental goods and c) the analysis of different procedures to introduce several biodiversity measures into a forest management optimisation model. These three significant scenarios demonstrate the importance of using methodological approaches underpinned by decision analysis tools.

Keywords: Environmental Economics; Decision Analysis; Forest Management; Biodiversity; Goal Programming; Compromise Programming

1. Introduction

The problems addressed by the discipline known as Environmental Economics are very complex, for at least the following two reasons:

a) Environmental assets are managed within joint production schemes, providing two sets of outputs. One set of outputs is formed by private goods, which are sold in markets. The other set is made up of public goods/bads for which there are not defined markets. It is well known in economic analysis that the presence of public goods/bads (externalities) generates a divergence between the private and the

* Departamento de Economía y Gestión Forestal, ETS Ingenieros de Montes, Universidad Politécnica de Madrid, Avenida Complutense s/n, 28040 Madrid, Spain. Fax: 34-91-5439557, Email: auggiegp@montes.upm.es

social optima (e.g., Varian 1984, chapter 7). This divergence implies a Pareto inefficiency in the allocation of resources (market failure) that should be removed in one way or in another.

b) Any decision taken in the environmental field affects several criteria of very different nature: economic, ecological, social, etc. Moreover, these criteria are usually in conflict. Hence, for the optimum design of environmental management policies, the decision-maker -private or public- does not aim to maximise a well-defined single objective function, as traditional analysis assumes.

In the last three decades, microeconomics in general and cost-benefit analysis in particular have extended their scope to accommodate the aforementioned problems. The overall idea consists in measuring the increase or decrease of social economic welfare due to the enjoyment/suffering of environmental public goods/bads through increases/decreases in the corresponding consumer and producer surpluses. This approach requires a previous estimation of demand and supply functions for the corresponding environmental commodities. Since there are generally no markets for the environmental commodities, demand and supply functions have to be estimated with the help of specific methods. These methods create virtual or contingent markets or estimate the influence of a certain amount of good/bad on the price of a private good with a well defined market (see Smith 1996 for a rigorous presentation of these methods).

The methods mentioned have been applied to several real problems with a good level of success. For instance, in the appraisal of recreational benefits of a natural area (e.g., Marinelli et al., 1990). However, the foundations for these methods are not exempt of difficulties. The main problem arises from the necessity, inherent to these methods, of reducing all the benefits and costs associated to the use of an environmental asset -with its undoubtedly multidimensional character- to a single monetary figure. This figure measures the net aggregation of increases and decreases in the corresponding surplus caused by a given environmental improvement or damage. This kind of analysis works reasonably well sometimes, however, in certain cases these methods require very strong assumptions demanding precise information, which is very difficult to obtain in practice.

This paper presents an overview of recent results where the above environmental problems are addressed in a different way. The basic idea is to demonstrate how decision analysis, chiefly in its multicriteria version, can be a very powerful tool in environmental economics. Thus, in Section 2 the appraisal of environmental assets is addressed in a non-conventional way by explicitly recognising the multicriteria conflict between the performance of environmental and economic assets and by using shadow values instead of using market prices. In this way, a non-monetary cost-benefit analysis is proposed. In Section 3, the divergence between the private and social optima due to the existence of public goods/bads is analysed by combining ideas from traditional microeconomics and modern decision analysis. Finally, Sections 4 and 5 focus on how to incorporate a measure of biodiversity into a forest management optimisation model, illustrating how different multicriteria decision methods can be sound devises for addressing this crucial problem.

2. A Shadow Appraisal of Environmental Assets

This line of research aims to develop a methodology applicable to the appraisal of environmental assets. The methodology takes into account the following two basic aspects:

a) The usual conflict between the performance of environmental and economic assets.
b) The values estimated for the environmental assets are not market values but shadow or internal values. These are the lowest prices capable of covering the inherent production costs.

The proposed methodology can be especially useful for measuring the relative efficiency, through non-monetary cost-benefit analysis, of potential investments in natural areas. Moreover, the approach also represents a useful first step in the optimal social provision of environmental public goods discussed in the next section.

Let us assume the following general setting. Vector $x_1, \ldots, x_i, \ldots x_m$ represents the amount of outputs provided by a public natural area. Without loss of generality, it is assumed that all the outputs enjoy the property "more is better". For the current resource level k we have the following transformation curve:

$$T(x_1, \ldots, x_i, \ldots, x_m) = k \qquad (1)$$

The above curve or frontier defines the technological domain of the problem. That is, the feasible and efficient mixes of outputs that can be obtained for a level of resources k that imply an aggregate cost R. The usual assumptions of convexity, continuity and differentiability of (1) are accepted.

The following family of iso-utility or welfare functions is introduced:

$$U(x_1, \ldots, x_i, \ldots, x_m) = \lambda \tag{2}$$

The usual assumptions of increasing monotonicity and concavity of (2) are accepted. The optimal mix of outputs $(x_1^o, \ldots, x_i^o, \ldots, x_m^o)$ will be obtained by maximising utility function (2) subject to transformation curve (1).

In order to determine the value associated to this optimal mix, we need to attach a vector of weights $(w_1, \ldots, w_i, \ldots, w_m)$ to the optimal mix of outputs. Within our context, market prices should not be used as weights for two reasons: a) market prices are not related to the production scenario and b) there is no actual market prices for many of the outputs provided by a natural area due to their character of public goods. However, it seems sensible to conceptualise weights as shadow or internal values within our context. Hence, the expression

$\sum_{i=1}^{m} w_i x_i^o$ measures the aggregate shadow values of the optimal mix.

This aggregate shadow value can be used, among other things, to construct ratios cost-benefit as illustrated at the end of this section. Let us now address the way to determine the vector of shadow weights. Within our context, these weight-shadow values should hold the following two conditions (Ballestero and Romero, 1993, 1998 chapter 7):

1º) The shadow value $\overline{R}$ or shadow revenue of every mix must be greater than or equal to the aggregate cost R of resources invested in the joint production process (i.e., R as it was defined before, represents the cost associated to the production of mixes on the frontier T = k). In fact, if this condition is not fulfilled the shadow revenue does not cover production costs what is economically untenable. Therefore we have the following condition:

$$\bar{R} = \sum_{i=1}^{m} w_i x_i \geq R \tag{3}$$

2º) The difference between the shadow values of the mix $\bar{R}$ and the aggregate cost R of the joint production process has to be as small as possible in order to avoid any overestimation. In fact any overestimation of the aggregate shadow value does not seem to be a justifiable policy for the policy maker. The two conditions 1º) and 2º) brought together lead to the following optimisation problem:

$$\mathrm{Min}\left(\bar{R} - R\right) = \mathrm{Min} \sum_{i=1}^{m} w_i x_i$$

subject to: (4)

$$\bar{R} = \sum_{i=1}^{m} w_i x_i \geq R$$

For a convex frontier, the optimisation problem given by (4) has the following unique solution (Ballestero and Romero 1993):

$$w_i = \frac{R}{\left(x_i^* - x_{i^*}\right)\left(1 + \sum_{i=1}^{m} \frac{x_{i^*}}{x_i^* - x_{i^*}}\right)} \tag{5}$$

where the vector $x_1^*,\ldots,x_i^*,\ldots,x_m^*$ represents the anchor values or ideal point. That is, the maximum for each individual output over the frontier T = k. The vector $x_{1*},\ldots,x_{i*},\ldots,x_{m*}$ represents the anti-ideal values or nadir point. That is, the worst possible value for each possible output on the frontier T = k.

The next step in the analytical procedure will consist in determining the optimal mix of outputs $\left(x_1^o,\ldots,x_i^o,\ldots,x_m^o\right)$. The conceptual and operational difficulties associated to the determination of a reliable representation of the utility or welfare function are enormous. For these reasons, resorting to a compromise approach will approximate the optimal mix or social equilibrium. Hence, the following general compromise programming model is formulated (Yu 1973, Zeleny 1974):

$$\mathrm{Min}L_p = \left[\sum_{i=1}^{m} w_i^p \left(x_i^* - x_i \right)^p \right]^{\frac{1}{p}}$$

Subject to: (6)

$$T(x_1, \ldots, x_i, \ldots, x_m) = k$$

where shadow weights w_i have already been defined (see expression (5)), and p is a parameter (metric) defining the family of distance functions. Compromise model (6) attempts to obtain the nearest point (or portion) of the frontier T = k with respect to the ideal point. This kind of compromise solution holds useful properties such as: feasibility, least group regret, Pareto optimality, independence of irrelevant alternatives, etc (Yu, 1985, pages71-74). Moreover, Yu (1973) demonstrated that for problems with two criteria (outputs in our context), the p = 1 and p = ∞ metrics define a subset of the transformation curve or frontier called *compromise set*. The other best-compromise solutions fall between the solutions corresponding to both metrics. Blasco et al. (1999) extended the boundness of the compromise set for a general case with m criteria (outputs) under very general conditions. Basically the only specific assumption required by Blasco et al. is the differentiability of the transformation hypersurface. Therefore, by solving (6) for metrics p = 1 and p = ∞ a compromise set as a closed set or portion of the frontier T = k nearest with respect to the ideal point is obtained.

For easy reading, the case corresponding to nil anti-ideals will be chosen (i.e., $x_{i*} = 0 \ \forall i$). In this situation, the following optimisation problem is obtained by making p = 1 in (6):

$$\mathrm{Max} \sum_{i=1}^{m} \frac{x_i}{x_i^*}$$

Subject to: (7)

$$T(x_1, \ldots, x_i, \ldots, x_m) = k$$

The above solution implies the maximisation of a separable and additive utility function $U(x_1+\ldots+x_i+\ldots+x_m)$ such as linear structure $U = k_1x_1+\ldots+k_ix_i+\ldots+k_mx_m$. Therefore, the bound L_1 of the compromise set corresponds to a solution of maximum efficiency.

The other bound L_∞ of the compromise set is obtained by making $p = \infty$ in (6). It is demonstrated (Ballestero and Romero 1991) that this substitution leads to the following system of m equations with m unknowns:

$$\frac{x_1}{x_1^*} = \ldots = \frac{x_i}{x_i^*} = \ldots = \frac{x_m}{x_m^*} \tag{8}$$

$$T(x_1, \ldots, x_i, \ldots, x_m) = k$$

The solution given by (8) represents balanced allocations between the m outputs considered. It is also interesting to point out that the $L\infty$ bound implies the maximisation of a Rawlsian utility function where the m outputs considered are perfectly complementary (see Tamiz et al. 1998 for technical details).

Let us represent by $x_1^1,\ldots,x_i^1,\ldots,x_m^1$ the vector of outputs corresponding to the L_1 bound of the compromise set and by $x_1^\infty,\ldots,x_i^\infty,\ldots,x_m^\infty$ the vector of outputs corresponding to the $L\infty$ bound of the compromise set. By taking into account both vectors, a good estimator of the aggregate shadow value for the natural area considered belongs to the closed interval $[R_1, R_\infty]$, being R_1 and R_∞ equal to:

$$R_1 = \sum_{i=1}^{m} w_i x_i^1 = R\sum_{i=1}^{m} \frac{x_i^1}{x_i^*} \tag{9}$$

$$R_\infty = \sum_{i=1}^{m} w_i x_i^\infty = R\sum_{i=1}^{m} \frac{x_i^\infty}{x_i^*} \tag{10}$$

The approach presented above can be easily adapted to the valuation of environmental improvements or damages. Thus, suppose that some improvements are undertaken in the natural area. These improvements shift the frontier $T = k$ in the north-east direction. This shift of the frontier implies a new cost R^N associated to the production of mixes on the new frontier as well as a new vector of anchor values $\bar{x}_1^*,...,\bar{x}_i^*,...,\bar{x}_m^*$. This new vector of anchor values implies new bounds for the compromise set as well as a new vector of shadow weights $\bar{w}_1,...\bar{w}_i,...,\bar{w}_m$. Therefore the shadow values of the environmental improvement is comprised within the following values :

For metric $p = 1$
$$\Delta R_1 = \sum_{i=1}^{m} \bar{w}_i \bar{x}_i^1 - \bar{R}_1 \tag{11}$$

For metric $p = \infty$
$$\Delta R_\infty = \sum_{i=1}^{m} \bar{w}_i \bar{x}_i^\infty - \bar{R}_\infty \tag{12}$$

If we represent the difference between the costs associated to the production of mixes in the new frontier with respect to the first frontier by ΔR (i.e., $\Delta R = R^N - R$), then the value of the cost-benefit ratio of the environmental improvement according to (11) and (12) belongs to the following closed interval:

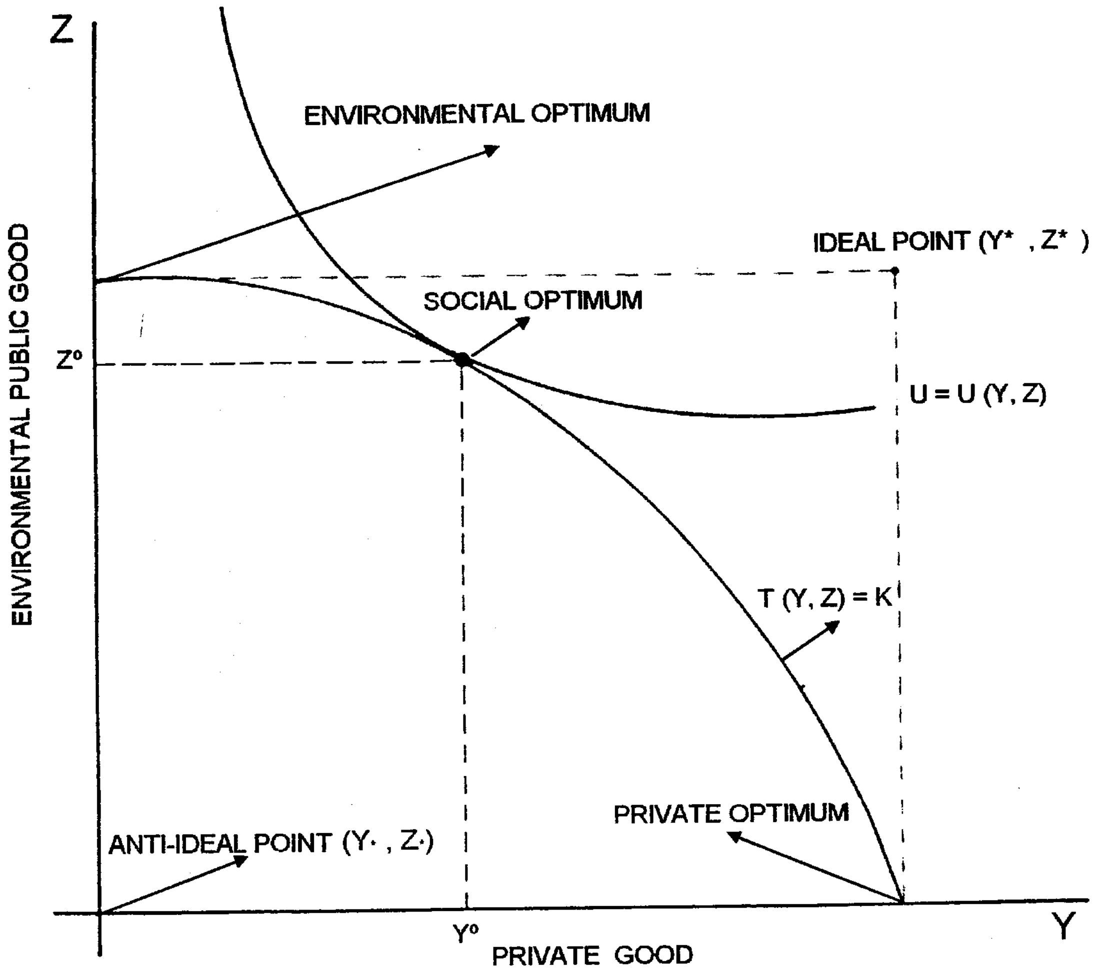
$$\left[\frac{\Delta R_1}{\Delta R}, \frac{\Delta R_\infty}{\Delta R}\right]$$

The approach expounded here can present certain advantages with respect to the traditional environmental valuation methods (travel cost, contingent valuation, hedonic prices, etc). Thus, instead of monetising the value of the environmental improvement by resorting to the creation of surrogate markets or the use of hedonic variables, the monetary value of the cost of the improvement is related to the shadow value increase of the environmental good. This approach can be especially useful when we want to compare the relative efficiency of several natural areas that provide a set of economic as well as environmental goods (see Ballestero 1999).

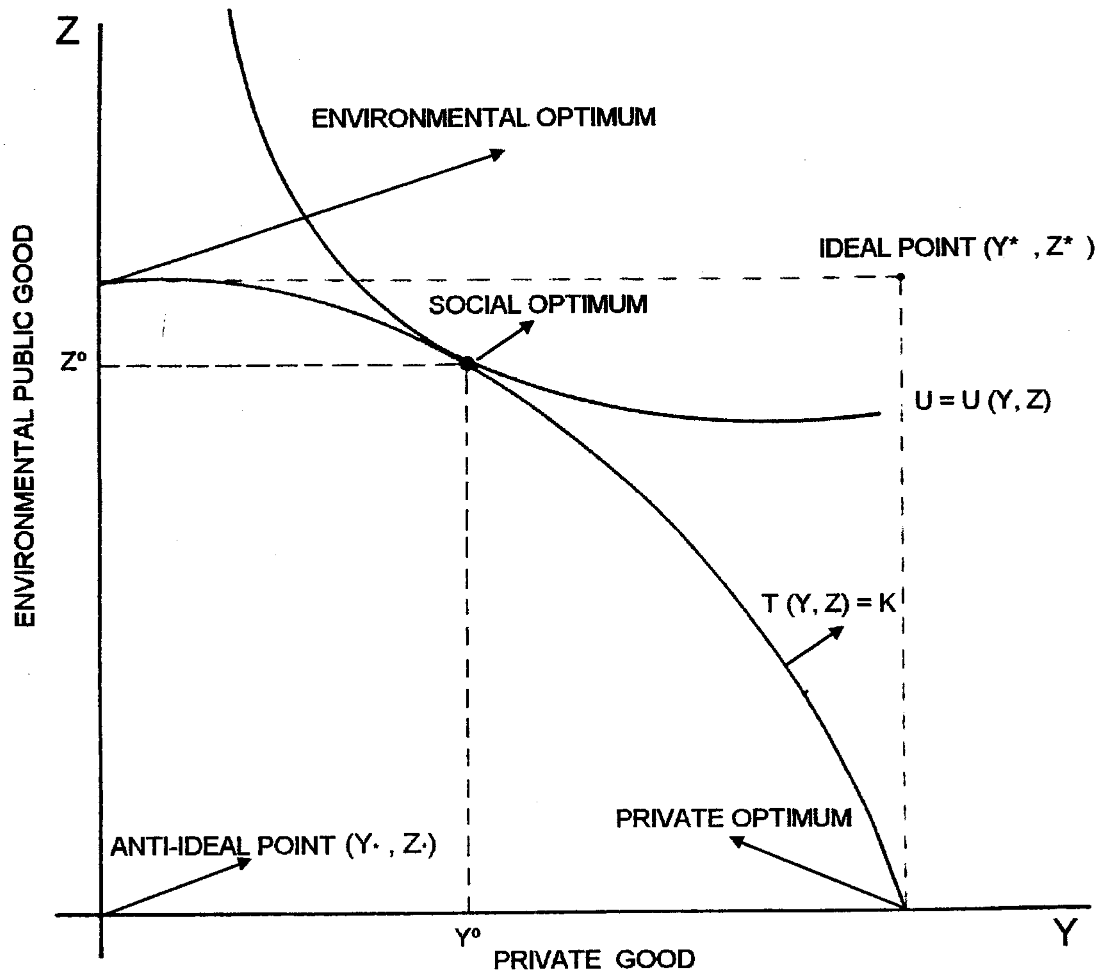
3. Optimal Provision of Environmental Public Goods

Let us consider the case of a farmer producing a private good sold in a market and an unpriced environmental public good. We represent the amounts produced of private and public good by Y and Z, respectively and the price of the private good by P_Y. For the current

ERRATA CORRIGE DELLA Fig. n.1 di pag. 19



ERRATA CORRIGE DELLA Fig. n.1 di pag. 19



resource level, the farmer faces a production possibility frontier or transformation curve in the Y-Z space given by:

$$T(Y, Z) = K \tag{13}$$

Figure 1 represents the above function, showing its typical concavity towards the horizontal axis. Some authors (e.g., Webster, 1999) have suggested that for low levels of the private good there is a certain complementary relationship between the production of the two goods.

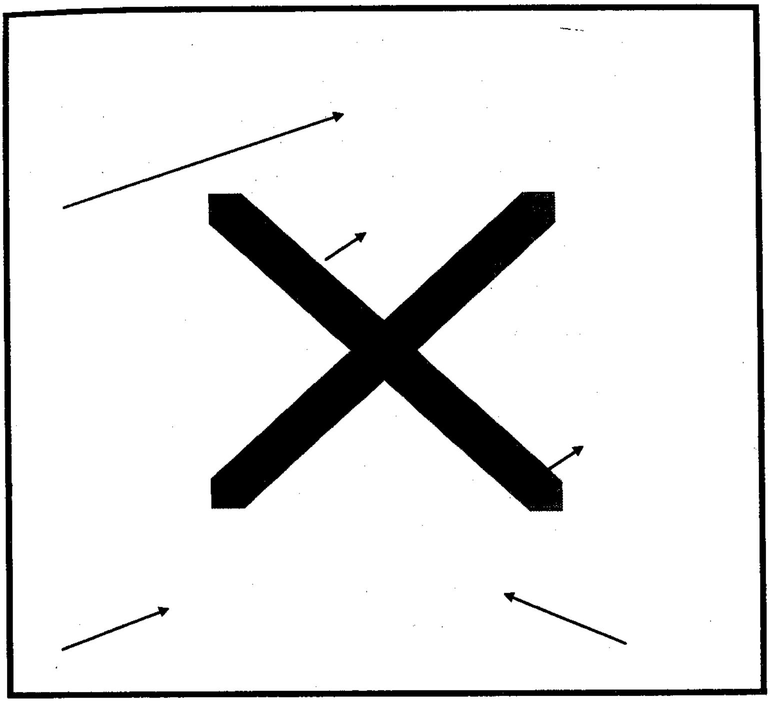


FIGURE 1. PRODUCTION POSSIBILITY FRONTIER, PRIVATE OPTIMUM, ENVIRONMENTAL OPTIMUM AND SOCIAL OPTIMUM.

The utility or welfare of the society is defined by the following function:

$$U = U(Y, Z) \tag{14}$$

The above social welfare function enjoys the usual properties (i.e., $U_Y \geq 0$, $U_Z \geq 0$, $U_{YY} \leq 0$, $U_{ZZ} \leq 0$). By taking into account this setting, the following three optima are derived:

The environmental optimum. It is the point where the production of the environmental public good achieves a maximum value. This optimum Z^* corresponds to the point where $T(Y, Z) = K$ intersects the Z-axis (see Figure 1).

The private optimum. It is the point where the revenue of the farmer achieves a maximum. Since the farmer does not receive any compensation for the public good produced, this optimum value Y^* corresponds to the point where $T(Y, Z) = K$ intersects the Y-axis (see Figure 1).

The social optimum. It is the point where the social welfare function $U(Y, Z)$ reaches a maximum value over the frontier $T(Y, Z) = K$. This optimum value (Y^o, Z^o) corresponds to the point of tangency between the iso-utility map $U(Y, Z) = l$ and the frontier $T(Y, Z) = K$ as shown in Figure 1.

The above analysis leads us to an important conclusion: the production of an environmental public good in competition with a private good generates a divergence between the private and the social optima. This divergence implies that the allocation of resources provided by the market is inefficient. In fact, in the private optimum the environmental good is under-produced in $Z^o - Z_*$ units, while the private good is over-produced in $Y^* - Y^o$ units.

One possible way to remove this inefficiency will consist in providing the farmer with a premium or subsidy P_Z for each unit of environmental public good produced. In this case, the objective function of the farmer would be given by:

$$\text{Max } P_Y Y + P_Z Z \tag{15}$$

The first term of (15) represents the benefits of the farmer as a producer of a private good provided by the market. The second term of (15) represents the benefits of the farmer as a producer of an environmental good provided by the society as a compensation for the positive externality generated by the farmer. By maximising objective function (15) over the frontier $T(Y, Z) = K$, the following well-known first-order condition is obtained (see e.g., Henderson and Quandt, 1958, pages 89-96):

$$\frac{P_Y}{P_Z} = \frac{T_Y}{T_Z} \qquad (16)$$

where the quotient of partial derivatives T_Y/T_Z represents the marginal rate of transformation between the private and the environmental good. By substituting P_Y by its observed value in (16) and by calculating the partial derivatives T_Y and T_Z in the social optimum (Y^o, Z^o) the following subsidy P_Z is obtained:

$$P_Z^* = P_Y \,(T_Z/T_Y)_{(Y=Y^o \;\; Z=Z^o)} \qquad (17)$$

The above subsidy is optimum because it allows for the restoration of the social efficiency in the allocation of resources. It is important to note that the proposed approach can be easily extended to a general setting where n private goods and m environmental public goods are considered. However, we should be aware that the implementation of the procedure requires the previous determination of the social optimum, something, which is not an easy task. The determination of a reliable representation of a welfare social function involving as arguments private and environmental goods is a very difficult task. Consequently, a pragmatic approach will consist in formulating a Compromise Programming model to determine the compromise set as a good surrogate of the social optimum, as it was expounded in the preceding section. An adaptation of this approach to a real case where the timber produced is the private good and the CO_2 captured by the forest is the environmental public good can be seen in Romero et al. (1998).

To illustrate how the proposed procedure works let us suppose the following equation for the frontier or transformation curve:

$$(Y\text{-}3000)^2 + 9(Z\text{-}1000)^2 = 36000000 \qquad (18)$$

From (18) the following anchor and nadir values are straightforwardly obtained:

$$Y^* = 9000 \quad Z^* = 3000$$

$$Y_* = 3000 \quad Z_* = 1000$$

It is also very simple to derive the following two optimum points from (18):

Environmental optimum: $Y^E = 3000 \quad Z^E = 3000$

Private optimum: $Y^P = 9000 \quad Z^P = 1000$

By applying Compromise Programming, the two bounds of the compromise set L_1 and L_∞ coincide and are equal to (Romero 1997):

$L_1 \rightarrow L_\infty \quad Y^o = 7243 \quad Z^o = 2414$

The above compromise is considered the surrogate of the unknown social optimum. The market solution (private optimum) implies an under-provision of the environmental good of $Z^o - Z_* = 2414 - 1000 = 1414$ units and an over-provision of the private good of $Y^* - Y^o = 9000 - 7243 = 1757$ units. Let us assume that the price P_Y of the private good is 30 Euros. For this situation, one feasible policy measure to obtain an efficient allocation of resources will consist in establishing a governmental subsidy P_Z^* per unit of environmental public good produced equal to (see expression (17)):

$$P_Z^* = 30\left[\frac{18(Z-1000)}{2(Y-3000)}\right]_{(Y=7243,Z=2414)} = 90 \text{ Euros}$$

It is easy to check that if we maximise the new objective function $30Y + 90Z$ over the frontier the surrogate of the social optimum ($Y^o=7243$, $Z^o=2414$) is obtained.

4. Forest Management Optimisation Models and Biodiversity: A Review

Society demands from forests public goods and services, including the protection of biodiversity . From the point of view of forest management, the most promising concepts of biodiversity are species diversity and habitat diversity. In this section, some of the major efforts undertaken in the last few years to incorporate these two biodiversity views into forest management will be reviewed.

The works by Kangas and Kuusipalo (1993), and Kuusipalo and Kangas (1994) represent an old line of research in the above direction. The general purpose of these studies was to develop a method capable of considering biological diversity as "another objective" in strategic forest management.

The Analytic Hierarchy Process (AHP) developed by Saaty (1980, 1994) is the optimisation tool used by Kangas and Kuusipalo to undertake their task. The choice problem is represented with the help of a decision hierarchy. The hierarchy consists of several levels of decision-making. The first level of the hierarchy defines the main purpose of the decision problem, namely, to maximise the overall utility of the forest area being planned. In the second level, the decision criteria taken into account in the planning process are added. In these studies, these are mainly timber production and biodiversity. In subsequent levels of the hierarchy, the general criteria are broken down into more detailed sub-criteria. Thus, the timber production criterion is usually broken down into sub-criteria such as the net incomes in the different periods considered, and the monetary value of the growing stock at the end of the planning horizon. In a similar way, the biodiversity criterion is broken down into different components considered relevant by the decision-maker. Kangas and Kuusipalo (1993) considered the richness, rarity and vulnerability of species as the dimensions of biological diversity for the studied forest area. Kuusipalo and Kangas (1994) considered as the most important dimensions of biological diversity in the forest area the following: the species dependent on the existence of old forest, the species dependent on the young forest, and the species dependent on the presence of hardwood trees.

The result of the process is an overall relative priority index for each feasible alternative. The relative index is a linear priority function obtained by multiplying the sub-priorities by the relative weight of each component and then adding up all the partial priorities. As a result, a ranking of the set of feasible alternatives is obtained. The best alternative of the ranking is the solution for which the overall utility is maximised according to the preferential weights placed on each criterion and sub-criteria by the decision-maker.

For instance, in Kangas and Kuusipalo (1993) the following biodiversity index was obtained:

$$BI_i = 0.12P_{iS} + 0.39P_{iR} + 0.49P_{iv}$$

where:

BI_i = biodiversity index for ith alternative.

P_{iS} = priority of ith alternative with respect to species richness.

P_{iR} = priority of ith alternative with respect to rarity.
P_{iv} = priority of ith alternative with respect to vulnerability.

It is rather obvious that to this kind of approach underlies a strong separability among the three criteria considered which implies the acceptance of an assumption of mutual independence of preferences (see Keeney and Raiffa, 1976). Within a biodiversity context, this can be very strong assumption.

The ranking of alternatives obtained with this procedure depends on the weights attached to the different criteria and sub-criteria. Moreover, for another planning problem, not only the weights but also the criteria chosen as representative of the biological diversity can be different, depending also on the scale of the management unit. For these reasons, the result of the process is specific for each planning problem and cannot be generalised.

The determination of the weights to be attached to the criteria and sub-criteria into which they are broken down can be regarded as a political problem concerning different decision-makers (landowners, politicians, citizens, etc). As pointed out by the authors, the main disadvantages of the approach are the lack of general agreement with respect to an operational definition of biodiversity and a lack of objective knowledge to quantify the alternatives with respect to their effect on the dimensions of the biological diversity chosen. The number of alternatives that can be added to the lowest level of the hierarchy is also limited given that the priorities of the alternatives are obtained by means of pairwise comparisons. However, the authors sensibly state that the choice made with the help of this approach can be considered as a good starting point for tactical and operational planning.

The papers of Kangas and Pukkala (1996), Pukkala et al. (1997) and Kangas et al. (1998) represent another direction for tactical planning closely related to the above. In these studies, certain variables relevant to species diversity and quantifiable in the inventories at the level of trees and stands are chosen as indicators of biodiversity for the forest area. Thus, Kangas and Pukkala (1996) considered the following indicators: the mean volume of deciduous trees, the percentage or proportion of old forest area and the mean volume of deadwood. The chosen components can also be broken down into more detailed categories such as different stages of decomposition of deadwood, volume of different species of broad-leaved trees, different age classes of old forest, etc.

The individual utility functions are determined as follows. First, several treatment schedules per compartment for the planning horizon are generated by computer simulation. The simulation exercise predicts the future development of the characteristics of the stand or compartment resulting from the implementation of a certain treatment schedule. Given the huge number of generated treatment schedules, it is impossible, in practical terms, to apply the AHP method to determine the relative priorities of these schedules with respect to the components of the objectives. This difficulty is solved with the help of the following process. The maximum and minimum values of a component are determined by simulating all the treatment schedules of the compartments. The interval between those values is divided into a "small" number of intervals of equal width. Then, the AHP method can be applied by taking the boundaries of two intervals at a time, and determining their relative importance or preference with respect to the level of achievement of the corresponding component. In this way, the sub-priorities or partial utilities of all the treatment schedules can be obtained by linear interpolation in terms of the sub-priorities corresponding to the amount of component they produce.

The overall utility function for any treatment schedule is now determined by multiplying the relative sub-priority functions by the relative weights of the objectives or components and adding the corresponding individual functions of every component. The relative weights are computed by using the AHP method again. These weights reflect the relative importance of the objectives given by the decision-maker or by an expert in the case of the biodiversity objective. Once the overall utility function has been estimated, a heuristic optimisation method called HERO is used to maximise it. HERO searches for the optimum value of the overall utility function, through all the possible combinations of treatment schedules for the compartments.

Pukkala et al. (1997) attempt to improve the study of Kangas and Pukkala (1996) by dividing or separating the indicators of biodiversity into more detailed categories and taking into account two additional indicators of species diversity: the within-stand and between-stand variety in the ecosystem. They also point out the use of the Delphi technique, when several decision-makers have to define the biodiversity indicators and their corresponding weights.

Other studies (Spellerberg and Sawyer, 1995, 1996) aim at the establishment of biodiversity standards to apply in the management of forest areas. These standards are levels of quality that should be established with respect to each identified way of restoring or maintaining biodiversity in a forest area. That is, the standard is the target referring to a specified criterion or indicator of biodiversity towards which managers should strive.

Some examples of suggested standards are the following: to supply a certain volume of deadwood, to require a certain percentage of broad-leaved species in a conifer forest, to set aside from timber production a given width of buffer strip on each side of forest streams, etc. From the point of view of decision analysis, this approach implies to formulate the biodiversity standards as rigid constraints. This strategy means the acceptance of a lexicographic ordering of preferences between the economic objective and the biodiversity standard (no finite trade-off between economic performance and biodiversity), which can be unrealistic (see Romero, 1991, pages 43-47).

Finally, another line of research is presented in the studies of Buongiorno et al. (1994, 1995). The level of analysis of these studies is the forest stand managed with selective cutting. These studies assume that the species diversity of a forest stand is strongly related to its structural diversity. The structural diversity of a stand is described by means of the distribution of trees by species-size classes, and the Shannon-Wiener index is used to quantify the proportional or relative abundance of trees in each species-size class.

The modelling framework consists of a mathematical programming structure with two objectives: an indicator of economic performance and the Shannon index measuring the structural diversity of the forest stand. In this way, the trade-offs between structural diversity and the economic objective are determined.

The formulation of the model requires the previous estimation of a growth sub-model for the stand for a specified time interval. The growth sub-model consists of a matrix **G** and a vector **c**. Matrix **G** includes the following items: 1) the probabilities that the trees in each species-size class remain in the same category, 2) the probabilities of transition from one species-size class to the next one and 3) the coefficients representing the influence of the trees in each species-size class on the ingrowth. The coefficients of vector **c** represent the part of the ingrowth independent of the stand state.

The approach by Buongiorno *et al.* is refined by Önal (1997a, 1997b), by proposing the use of what he calls the "normalised absolute deviation index"(NAD index), instead of using the Shannon index. The NAD index measures the divergence between the relative abundance of trees in each species-size class of the stand and the corresponding relative abundance of an arbitrarily specified target distribution that represent the desired stand structural diversity composition. The mathematical formulation of the optimisation model can be expressed in the following simplified way:

Objective function:

$$\text{Max NPV} \tag{19}$$

Constraints:

$$\mathbf{y} = f(\mathbf{h}, \mathbf{G}, \mathbf{c}, \mathrm{w}) \tag{20}$$

$$2(1-\beta) - \frac{\sum_{i=1}^{n} \left| y_i - \beta_i \sum_{i=1}^{n} yi \right|}{\sum_{i=1}^{n} y_i} \geq \alpha \tag{21}$$

$$\mathbf{h} - \mathbf{y} \leq \mathbf{0} \tag{22}$$

where:

NPV = net present value of the harvest over an infinite horizon.

$\mathbf{y} = (y_i)$ = vector of steady-state tree stock by species-size class.

$\mathbf{h} = (h_i)$ = vector of steady-state harvest by species-size class.

G and **c** = matrix and column vector, respectively of the growth sub-model that were defined above.

w = mortality rate during harvesting.

β_i = specified proportion or relative abundance of the ith class in the specified target distribution, $\{b_i\}$.

$\beta = \min\{b_i\}$

α= exogenous specified level of structural diversity, constraining the divergence between the proportions in each species-size class and the proportions of the target distribution.

In order to understand the functioning of the model, it is important to note the following facts: a) Equation (20) describes the stand growth and steady-state condition; b) The left-hand side of constraint (21) represents the normalised absolute deviation index, while the right-hand side, a, is an arbitrary parameter; c) Constraint (21) is non-linear but it can be linearised with the help of Goal Programming techniques (see Önal 1997b for details) and d) Constraint (22) is imposed to secure that harvest variables by species-size class cannot exceed the available stock variables.

From a mathematical programming point of view, model (20)-(22) corresponds to the constraint method of multiobjective programming (see e.g., Romero and Rehman 1989 pp.71-72). Through parametric variations of the right-hand side a of (21), the efficient set or production possibility frontier for the economic and structural diversity objectives is obtained. The slopes in the different points of the frontier measure the marginal rate of transformation between economic returns and structural diversity. Again, with the help of Compromise Programming it is possible to determine the portion of the frontier nearest with respect to the ideal point (α^*, NPV^*). This compromise set can be again considered a good surrogate of the utility optimum.

5. Forest Management Optimisation Models and Biodiversity: An Analytical Procedure

In this section an analytical framework for incorporating some ideas regarding biodiversity will be presented. These ideas have been taken from forest ecology (see Hunter, 1990 for a summary). The basic points for this argument are the following:

a) If the spatial diversity of a forest is maintained, then the wildlife species diversity supported by the forest is also maintained.
b) The spatial diversity of a forest can be maintained by managing the age structure of the forest, which is determined by the ages of all the stands that the forest comprises.
c) Managing for the diversity of wildlife species requires providing for old forests, as they constitute an important habitat for many species.

In coherence with these basic points a sound framework of biodiversity will aim at achieving the following purposes:

(1) Creating old stands by lengthening the financial rotation age of the forest stands.
(2) Having the entire stand age classes represented in the forest.
(3) Maintaining a "balance" of age classes, in the sense of establishing the same number of stands per age class.

(4) Arranging the stands of different sizes and ages into a mosaic of great spatial heterogeneity.

Point (3) express to some extent a condition of area regulation, when the area of the harvest units is similar. Point (4) relies on what is known as edge effect, understanding the edge as the frontier between adjacent stands. The edge effect assumes that the diversity and abundance of many wildlife species is larger near an edge (Harris, 1984, Hunter, 1990). A feature of the edge effect is called the contrast, understanding by contrast the age difference between adjacent units. Some authors (see Hunter, 1990, pages 109-112) argue about the ecological rationale of maximising the edge contrast. However, other authors state that maximising the edge contrast may increase habitat fragmentation. This damaging effect could diminish if the size of the harvest units is large enough for interior species (Hof and Bevers 1998). Moreover, Harris and McElveen (1981) and Harris (1984) have shown that in the long-term two adjacent stands will have a maximum edge contrast if one of the stands is cut when the other one is half-way through the rotation age. This condition implies that the age difference between adjacent stands should be equal to half of the forest rotation age.

An important task will consist of formulating a harvest scheduling model with the following properties:

A) Maximisation of the edge contrast.
B) Creation of old stands by lengthening the financial rotation age.
C) Representation of all stand ages in the forest.
D) Maintenance of a "balance" of age classes, in the sense of establishing the same number of stands per age class.

The following zero-one Goal Programming model proposed by Bertomeu and Romero (1999) allows to obtain a harvest schedule that holds all the properties:

Achievement function:

$$\text{Min} \sum_{i=1}^{k} \left(u_i + v_i\right) + \sum_{j=1}^{A} \left(\alpha_i + \beta_j\right) \tag{23}$$

Goals and constraints:

$$n_l + p_l + u_l - v_l = R/2 \qquad l=1,2,\ldots,K \tag{24}$$

$$F_z - Fi + n_l - p_l = 0 \qquad \forall i,z \in I \tag{25}$$

$$\sum_{j=1}^{h}\left[\frac{t}{2} + (h-j)t\right]S_{ij} + (I_i + T)S_{i,h+1} - F_i = 0 \qquad i=1,2,\ldots,m \tag{26}$$

$$\sum_{j=1}^{h+1} S_{ij} = 1 \tag{27}$$

$$\sum_{i}\sum_{j} S_{ij} = 0 \qquad \forall i,j \in J \tag{28}$$

$$\sum_{i=1}^{m} S_{ij} + \alpha_j - \beta_j = q \qquad j = 1,2,\ldots,h \tag{29}$$

$$\sum_{i} S_{i,h+1} + \alpha_j - \beta_j = q \qquad \forall i \in L \qquad j=h+1,\ldots,A \tag{30}$$

$$S_{ij} \in \{0,1\} \qquad \forall i,j \qquad b_i \in \{0,1\} \qquad \forall_1 \tag{31}$$

$$\sum_{i=1}^{m}\sum_{j=1}^{h} NV_{ij} \geq NV \tag{32}$$

Where:

I_i = initial age of ith harvest unit.

F_i = final age of ith harvest unit.

t = time unit or time span.

T = planning horizon.

R = final forest rotation age.

h = T/t = number of cutting periods.

m = number of harvest units.

I = index set of pairwise adjacent units.

J = index set of pair of values i and j that imply cutting a unit below its maturity age.

A = number of feasible age classes.

q = number of harvest units belonging to each age class (i.e., q = m/A).

NV_{ij} = net present value attached to the harvest of the ith unit in the jth cutting period.

S_{ij} = binary (0/1) variable, that is $S_{ij} = 1$ if the ith unit is cut in the jth period, otherwise $S_{ij} = 0$.

Goals (24)-(25) in conjunction with the first term of the achievement function (23) guarantees as much as possible that the age difference between adjacent harvest units is half of the forest rotation age R. In this way, the edge contrast is maximised. Constraints (26)-(27) guarantees that the final ages of the m harvest units are logically feasible. Constraint (28) secures that no unit is harvested before it reaches its maturity age. Finally, goals (29)-(30) in conjunction with the second term of the achievement function (23) guarantees, as much as possible, that the number of harvest units belonging to each age class is the same.

It is important to note that through parametric variations of the right hand side NV of (32) the efficient set or production possibility frontier in the biodiversity quality-economic returns space can be established. It should also be noted that constraints (26)-(27) of the above zero-one Goal Programming model allows a maximum of one cut within the planning horizon. However, the model can be extended to a general context where the number of harvests of each unit within the planning horizon can be more than one. Technical details about the model, as well as an application to a Douglas-fir forest can be seen in Bertomeu and Romero (1999).

6. Concluding Remarks

The results presented in the paper demonstrate how the decision analysis paradigm, chiefly in its multicriteria version, can be a powerful tool to address a large range of environmental economic problems. To be more precise, it has been demonstrated throughout the paper how multicriteria approaches such as Goal Programming and Compromise Programming seem especially relevant for tackling an important number of environmental problems.

Despite the promising success of the cases reported in this paper, there is not claimed of superiority of the methods based on decision analysis with respect to the approaches based on traditional microeconomics (such as cost-benefit analysis) within the environmental field. This paper has tried to transmit a different message: "all the methodologies have their own limits and cost-benefit analysis is not an exception". Within a context of environmental economics, there are problem situations where methods based on cost-benefit analysis have shown an interesting problem-solving capacity. Nevertheless, in the same field there are other situational contexts, where the classic methods are of dubious applicability while the multicriteria decision analysis methods seem to be very powerful. Research efforts for developing and applying multicriteria decision analysis to environmental economics seem a crucial and attractive intellectual challenge.

Acknowledgements

This research was supported by "Comisión Interministerial de Ciencia y Tecnología (CICYT)" and "Consejería de Educación y Cultura de Madrid". We are grateful to Dr. Pedro Cifuentes for providing us with relevant literature on forestry diversity. Thanks are given to Drs. Luis Díaz-Balteiro and Valeria Ríos for their helpful comments. Christine Mendez checked the English language.

BIBLIOGRAFIA

BALLESTERO, E., 1999. Measuring efficiency by a single price system. *European Journal of Operational Research,* **115**:616-623.

BALLESTERO, E. and ROMERO, C., 1991.A theorem connecting utility function optimization and compromise programming. *Operations Research Letters,* **10**: 421-427.

BALLESTERO, E. and ROMERO, C., 1993.Weighting in compromise programming: A theorem on shadow prices. *Operations Research Letters,* **13**:325-329.

BALLESTERO, E. and ROMERO, C., 1998. *Multiple Criteria Decision Making and its Applications to Economic Problems.* Kluwer Academic Publishers, Boston.

BERTOMEU, M. and ROMERO, C., 1999. A harvest scheduling model within a biodiversity context. A zero-one goal programming approach. Paper presented a t the 15th Triennial Conference of the International Federation of Operational Research Societies (IFORS99), Beijing, China.

BLASCO, F., CUCHILLO-IBÁÑEZ, E., MORÓN, M.A. and ROMERO, C., 1999. On the monotonicity of the compromise set in multicriteria problems. *Journal of Optimization Theory and Applications,* **102**: 69-82.

BUONGIORNO, J., DAHIR, S., LU, H-C. and LIN, C-R., 1994. Tree size diversity and economic returns in uneven-aged forest stands. *Forest Science,* **40**: 83-103.

BUONGIORNO, J., PEYRON, J-L., HOULLIER, F. and BRUCIAMACCHIE, M., 1995. Growth and management of mixed-species, uneven-aged forests in the French Jura: Implications for economic returns and tree diversity. *Forest Science,* **41**: 397-429.

HARRIS, L.D., 1984. *The Fragmented Forest. Island Biogeography Theory and the Preservation of Biotic Diversity.* The University of Chicago Press, Chicago.

HARRIS, L.D. and MCELVEEN, J.D., 1981. Effect of forest edges on north Florida breeding birds. IMPAC Reports vol. 6, no. 4. Cited from Harris 1984.

HENDERSON, J. M. and QUANDT, R. E., 1958. *Micoeconomic Theory. A Mathematical Approach.* McGraw-Hill, Tokyo.

HOF, J. and BEVERS, M. 1998 *Spatial Optimization for Managed Ecosystems.* Columbia University Press, New York.

HUNTER, M.L., 1990. Wildlife, forests, and forestry. Principles of managing forests for biological diversity. Wildlife Department, College of Forest Resources, University of Maine, Orono.

KANGAS, J. and KUUSIPALO, J., 1993. Integrating biodiversity into a forest management planning and decision-making. *Forest Ecology and Management*, **61**: 1-15.

KANGAS, J. and Pukkala, T., 1996. Operationalization of biological diversity as a decision objective in tactical forest planning. *Canadian Journal of Forest Research*, **26**: 103-111.

KANGAS, J., ALHO, J.M., KOLEHMAINEN, O. and MONONEN, A. 1998. Analyzing consistency of experts' judgments-case of assessing forest biodiversity. *Forest Science*, **44**: 610-617.

KEENEY, R. L. and RAIFFA, H., 1976. *Decisions with Multiple Objectives: Preferences and Value Trade-offs*. John Wiley and Sons, New York.

KUUSIPALO, J. and KANGAS, J., 1994. Managing biodiversity in a forestry environment. *Conservation Biology*, **8** : 450-460.

MARINELLI, A., CASINI, L. and ROMANO, D. (1990). User benefits and the economic regional impact of outdoor recreation in a natural park of northern Tuscany, in *Land Use for Agriculture Forestry and Rural Development (Whitby, M. C. and Dawson, P.J.,* editors). Department of Agricultural Economics, The University, Newcastle upon Tyne, 179-193.

MEALEY, S.P., LIPSCOMb, J.P. and NORMAN, K.J., 1982. Solving the habitat dispersion problem in forest planning. *Trans. North* Am. *Wild. Nat. Resour. Conf.* **47**: 142-153. Cited from Hunter 1990, p. 112.

ÖNAL, H., 1997a. A computationally convenient diversity measure: theory and application. *Environmental and Resource Economics*, **9**: 409-427.

ÖNAL, H., 1997b. Trade-off between structural diversity and economic objectives in forest management. *American Journal of Agricultural Economics*, **79**: 1001-1012.

PUKKALA, T., KANGAS, J., KNIIVILÄ, M. and TIAINEN, A-M, 1997. Integrating forest-level and compartment-level indices of species diversity with numerical forest planning. *Silva Fennica*, **31**: 417-429.

ROMERO, C., 1991. *Handbook of Critical Issues in Goal Programming*. Pergamon Press, Oxford.

ROMERO, C., 1997. Multicriteria decision analysis and environmental economics: An approximation. *European Journal of Operational Research*, **96**: 81-89.

ROMERO, C. and REHMAN, T., 1989.*Multiple Criteria Analysis for Agricultural Decisions*. Elsevier, Amsterdam.

ROMERO, C., RÍOS, V. and Díaz-Balteiro, L., 1998. Optimal forest rotation age when carbon captured is considered: theory and applications. *Journal of the Operational Research Society*, **49**, 121-131.

SAATY, T.L., 1980. *The Analytic Hierarchy Process. Planning, Priority Setting, Resource Allocation*. McGraw-Hill, New York.

SAATY, T.L., 1994. How to make a decision: the analytic hierarchy process. *Interfaces* **24**: 19-43.

SMITH, V. K. 1996. *Estimating Economic Values for Nature. Methods for Non-Market Valuation*. Edward Elgar, Cheltenham.

SPELLERBERG, I.F. and SAWYER, J.W.D., 1995. Multiple-use, biological diversity and standards. *New Zealand Forestry*, February: 21-25.

SPELLERBERG, I.F. and SAWYER, J.W.D., 1996. Standards for biodiversity: a proposal based on biodiversity standards for forest plantations. *Biodiversity and Conservation*, **5**: 447-459.

TAMIZ, M., JONES, D.F., and ROMERO, C., 1998. Goal programming for decision making: an overview of the current-state-of-the art. *European Journal of Operational Research*, **111**:569-581.

VARIAN, H. R., 1984. *Microeconomic Analysis*. W. W. Norton and Company, New York.

WEBSTER, J. P. G., 1999. The challenge of sustainability at the farm level: presidential address. *Journal of Agricultural Economics*, **50**:371-387.

YU, P. L., 1973. A class of solutions for group decision problems. *Management Science*, **19**: 936-946.

YU, P. L., 1985. *Multiple Criteria Decision Making. Concepts, Techniques and Applications*. Plenum Press, New York.

ZELENY, M., 1994. A concept of compromise solutions and the method of the displaced ideal. *Computers and Operations Research*, **1**: 479-496.