
Biodiversity and Regional Sustainability Analysis

Daniel P. Faith

Division of Wildlife and Ecology, CSIRO, P. O. Box 84, Lyneham, A.C.T. 2602, Australia
phone: 61 62421683
fax: 61 62413343
e-mail: d.faith@cbr.dwe.csiro.au

ISBN 0 643 05764 1
CSIRO Australia 1995
Printed by Paragon Printers

Acknowledgements
I thank C. and W. Larmour for pointing out the fundamental difference between forgone forestry and foregone forestry. I thank S. Smith and R. Turner for help in preparation of this document, B. Foran for comments on the manuscript, and gratefully acknowledge the collaboration of P. Walker on the DIVERSITY package and associated methods for biodiversity assessment.

ABSTRACT

Evaluating sustainability at a regional level requires some assessment of the degree to which conservation and production criteria are integrated so as to provide high present and future net benefits to society. This assessment cannot be achieved simply by using indices based on comparisons for the individual criteria of observed indicator values with standards; not only may objective standards not be apparent but also the trade-offs among criteria may not be captured by such indices. A criterion of biodiversity protection presents special problems applied regionally, but recently has been incorporated into a trade-off or multi-criteria framework, avoiding the need for arbitrary standards or targets for biodiversity. Biodiversity and other criteria, such as forestry suitabilities, can be explored regionally by varying the relative weightings on the criteria and determining how these affect trade-offs and consequent total net benefits. Such trade-off spaces can be used to quantify regional sustainability - defined here as the degree to which a given region's particular capacity for trade-offs has been realised (or is potentially realisable). An example illustrates how two regions, having the same forgone biodiversity and same forgone forestry, can have different sustainability ratings, because the distribution of opportunities among the region's areas, and hence capacity for trade-offs, differs between the two.

Current sustainability of a region is quantified, for a given relative weighting of criteria, as a function of the difference between the total net benefit of the *current* land allocation/management, and the highest total net benefit that would have been achievable, *ignoring* the given land allocation/management and any current constraints on change. Potential sustainability, in turn, is defined as a function of the difference between the highest total net benefit still achievable *given* the current allocation plus constraints, and the highest total net benefit achievable again *ignoring* the current allocation/management and current constraints on change.

In the absence of any definitive fixed weightings on the basic criteria or objectives, sustainability must be acknowledged as not always reducible to one single number. However, sensitivity analysis may reveal that, over a range of weights, different land allocation scenarios consistently lead to different sustainabilities, or that some factors rather than others are consistently critical to achieving higher sustainability in a given region.

Using a simple example with two basic criteria, biodiversity and forestry, scenarios are illustrated in which

1) constraints on the set of areas given as already-protected versus already-forested dramatically affect current and potential regional sustainability,
2) land clearance reduces regional sustainability, but is partly compensated for by forest plantations on cleared land, and
3) a "sympathetic" form of forestry management of individual areas, providing partial biodiversity protection at a small cost, in some cases improves the overall sustainability rating of the region.

Introduction

This paper develops and illustrates a quantitative approach to the evaluation of "regional sustainability", with particular reference to the integration of biodiversity protection with other criteria, such as suitability of land for production or development. The term "regional sustainability" will be used here with the acknowledgement that it partly overlaps with, and partly contrasts with, the various definitions of "sustainable development" that have appeared since that term was popularised in the World Conservation Strategy ("WCS"; IUCN, 1980) and later in the "Brundtland report" (WCED, 1987). The WCS argued for the integration of conservation and development, promoting the idea that "conservation and sustainable development are mutually dependent". The World Commission on Environment and Development (WCED, 1987) subsequently defined sustainable development as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs". This definition has been restated since in a number of similar forms. In *Caring for the Earth* (IUCN, 1991), the successor to the WCS, the definition of sustainable development was restated as "improving the quality of human life while living within the carrying capacity of supporting ecosystems". Within the discipline of ecological economics (and elsewhere), sustainable development has been linked to a constrained form of the usual mechanisms for assessing economic development, as in the assertion: "it involves maximising the net benefits of economic development, subject to maintaining the services and quality of natural resources over time" (Pearce and Turner, 1990).

All these definitions reflect the integration of conservation and development as an essential element of sustainability, with a time-frame that includes future generations. However, the primary focus in each of the definitions stated above arguably is on development, with ecological/conservation concerns acting as a constraint or as a supporting consideration. Consequently, the definitions are sometimes interpreted to mean that, whatever else is done, we must sustain (maintain) the "natural resource stock" (e.g. Barbier, 1989; Pearce and Turner, 1990). Similarly, the goal may be taken to be the maintenance of development/production, but with the realisation that this goal is well-served only if attention also is given to the environment (e.g. Birch, 1993).

This asymmetry is not surprising given that these two different value systems inherently are hard to combine in any operational way that goes beyond basic definitions. This difficulty is reflected in the usual ways that conservation- and production-related criteria have been "integrated" to form overall measures of sustainability. For example, in *Caring for the Earth* (IUCN, 1991) it is argued that "to measure progress toward a sustainable society, we need indicators of quality of life and of ecological sustainability....by definition, indicators can measure only components of either". The list of possible indicators in that report illustrates a common approach to "integration"; a list of criteria is assembled, so that sustainability is taken to be indicated by the extent to which each of these individual criteria meets some standard or target. Integration is achieved by including on the list criteria relating to conservation and criteria relating to development. An overall measure of success may be derived by calculating a distance of the current state from a target or ideal situation, based on the identification of a target value for each of the individual criteria (for review, see Kuik and Verbruggen, 1991).

It will be useful to contrast the definitions and corresponding strategies noted above with other approaches to the assessment of sustainable development that share the "lists" perspective on combining conservation and production or other criteria, but also go beyond this in achieving true integration of conservation and development/production. These approaches consider the *trade-offs* among criteria and consequently the resulting *net benefits* for society.

In Australia, the link between sustainability, trade-offs and net benefits is highlighted in recent attempts to outline the requirements for sustainable development in the context of forestry:

"...improving material and non-material well-being will involve optimising the tangible and intangible social and economic benefits to the community attainable from forests. Achieving this will involve the development of policies and measures which address the following broad objectives:

the allocation of forest land to a mix of land uses according to the highest community value;

the optimisation of benefits to the community and long-term sustainability of all forest uses

- commercial benefits within ecological constraints
- intangible values and non-commercial uses consistent with achieving net optimum benefits; and
- the maintenance of options for future generations"

(Ecologically sustainable development working groups, 1991).

At present there is no accepted strategy for measuring and integrating the benefits referred to above, particularly considering benefits derived from protection of biodiversity. One explicit trade-off framework for sustainability that incorporates biodiversity is based on multi-criteria analysis (Munasinghe, 1993). Here, a trade-off space is defined and a surface of possible solutions then is identified (each of these possible solution is required to be "Pareto-optimal", as discussed below). While standards for some of the individual criteria may exist, the approach largely focuses on the best trade-offs among the criteria in the absence of individual standards. Achieving sustainability is implicitly equated with achieving a solution that lies on the Pareto-optimal surface (Munasinghe, 1993). Similar multicritieria approaches, although not linked to sustainability, are described in Nijkamp (1979) and Jansenn (1991). In all these approaches an efficient set of solutions is found, and then one of these solutions is selected.

Kangas and Kuusipalo (1993) and Kuusipalo and Kangas (1994) consider biodiversity criteria as part of a multicriteria framework but, like Munasinghe, only apply this criterion at the level of individual areas or projects. A recent extension of the multi-criteria analysis framework (Faith and Walker, in press b) that incorporates biodiversity at the regional level (where the overall biodiversity of sets of projects or areas must be taken into account) will be discussed below, following a brief review of other trade-offs approaches to sustainability.

Incorporating biodiversity into multicriteria analysis at a regional level raises the prospect of using this framework to evaluate regional sustainability. However, there has been little quantification of sustainability in the sense of regional trade-offs that might allow comparative evaluation of different regions, or monitoring of changes in sustainability of a given region. What is required is means of quantifying the degree to which a region, at any stage, has achieved (or is on a path towards achieving) its capacity for trade-offs. This paper proposes a framework for quantifying a form of "regional sustainability" applicable at the regional or higher level. Regional sustainability as defined here will reflect the region's success (or potential for success) in achieving effective trade-offs between conservation and development (or other criteria). This perspective is particularly appropriate for biodiversity assessments; it will be argued that there is generally no obvious way in practice to apply the constraint of "no loss", nor is there any obvious standard or target for regional biodiversity protection. However, the recent incorporation of assessments of the biodiversity of sets of areas into a trade-off or multi-criteria framework (Faith and Walker, 1994; in press b) provides a practical alternative approach.

While trade-offs are in principle applicable to the full range of different factors that have sometimes been listed as elements or indicators of sustainable development, this paper will use examples based on biodiversity protection as the primary conservation goal, with forestry as the primary development/production goal.

As with most cost-benefit or multicriteria methods, the values for various criteria may represent costs/benefits discounted over time, or represent total cumulative values predicted over some agreed time frame. In the examples presented here, forestry "opportunity" might be based on estimated values, for example, over a 50 year period.

Regional sustainability will depend on not only effective land-use allocations for alternative, sometimes competing, land uses, but also the degree to which management of a given area satisfies multiple goals. Regional sustainability therefore will include both regional trade-offs and trade-offs within a given area. In the examples presented here, the latter will correspond to partial protection of biodiversity and "sympathetic" forestry activity.

Regional biodiversity measures and their incorporation into a trade-offs framework will be briefly reviewed in the next section. Following this, the need for weights and consistent measures of distance is discussed in the context of alternative proposals relating to sustainability. This section will include discussion of the limitations of standards or targets. Finally, the principle of a region's capacity for trade-offs is introduced, along with corresponding quantitative measures of sustainability. Following this, properties of this family of regional sustainability measures will be illustrated and discussed using examples.

Biodiversity and trade-offs

Biodiversity protection constitutes an important special case among possible regional conservation goals. The reason is that *sets* of protected areas are to be characterised as having some (estimated) biodiversity, but this property of the set is not simply the sum of the biodiversity (e.g. the number of different species) contained within each member area. Because different areas within a region will overlap to some degree in their component species, the contribution of any one area to total biodiversity depends on which other areas are protected (the principle of complementarity, e.g. see Pressey et al. 1993).

The problem stated above is further complicated by the fact that biodiversity assessments must use some form of surrogate information in the absence of complete lists of species for different areas. Surrogate information can include environmental data (Pressey et al. 1993; Faith and Walker in press a). Here, the rationale is that the greater the environmental or habitat differences between two areas, the greater will be the degree to which their member species complement one another. Based on this rationale, and a robust model of species-environment relationships, an "environmental diversity" measure (ED, Faith and Walker, 1993; in press a) provides a mechanism for evaluating the (expected) relative number of species represented by different sets of areas chosen for protection.

As an example, suppose that 17 areas within a region are described by the environmental space of Fig. 1a. One dimension of this space might correspond to a rainfall index and the other to a temperature index. The ED measure takes complementarity among areas into account; for example, if area 9 is already protected, the additional protection of area 8 (which is very similar environmentally) is then expected to contribute only a relatively small number of additional species. If only three areas, for example, could be protected, expected biodiversity would be maximised (equivalently, "forgone biodiversity" would be minimised) according to the ED criterion if the three areas spanned the space better than any other set of three. The actual ED value, computed as the sum of the distances from all points in the space to their nearest protected area will be **low** when the represented biodiversity is **high**.[1]

[1] In practice, the boundaries of the realised environmental space, the relative species richness expected for different parts of the space, and other parameters, may be varied within the ED framework (Faith and Walker, 1993, 1994, in press a).

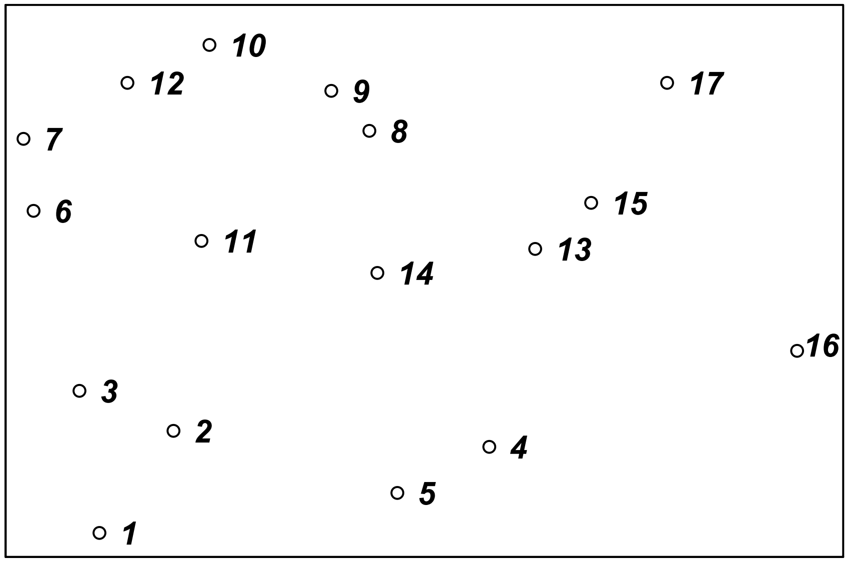

Figure 1 a) A hypothetical two-dimensional environmental space containing 17 areas.

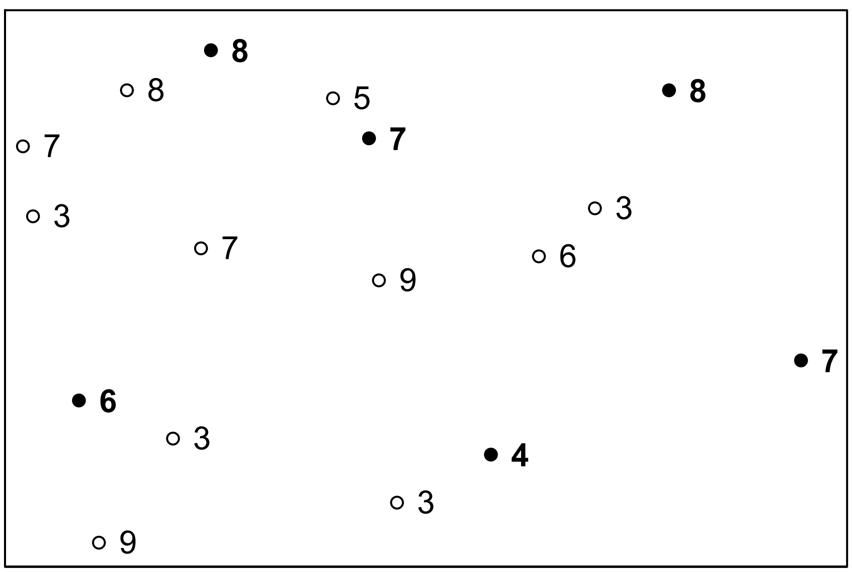

Figure 1 b) Labels for the 17 areas indicate relative costs of protection, equated here with forgone forestry opportunity. Six areas, shown in solid dots and bold labels, are protected. These six span the space well relative to other possible sets of six areas and have a forgone biodiversity index (ED) value of 365. The total cost of protection of these six areas is 40 units.

This biodiversity measure is open-ended in that each additional area will contribute something to represented biodiversity. Therefore, there is no obvious standard or level of adequacy of representation of biodiversity in protected areas; conceivably all areas might be protected for biodiversity, each making some unique contribution to the total. It follows that the level of protection adopted regionally may be determined in part by traded-offs with other competing land-use opportunities (Faith and Walker, 1994, in press b). This will have implications for measures of sustainability, as discussed below.

The measure of forgone biodiversity provided by ED (or any other surrogate biodiversity measure, see Faith and Walker, 1993, in press a) could be combined with a variety of other criteria as a basis for assessment of trade-offs at the regional level. A criterion corresponding to the estimated suitability of a given area for a competing land-use, forestry production, will be considered here. Allocation of an area to forestry production will be assumed (at least initially) to correspond to no protection of the biodiversity of that area. The forestry benefits derived from a given area might be defined, for example, as the total production value over a nominated time period, assuming a management regime compatible with sustained yield. Alternatively, the relative suitability value may reflect the area's status relative to a combination of forestry-related indicators or guidelines (for example, see Faith et al., 1994). An area's forestry suitability value then becomes an associated "cost" when the area is allocated to biodiversity protection (for an example, see Faith and Walker, 1994; Faith et al., 1994).

Examples using ED and forestry suitabilities are shown below. ED values and trade-offs with forestry opportunities are calculated using the DIVERSITY software package (Walker and Faith, 1993, 1994).

Weights and overall measures of sustainability

If we have a nominated set of protected areas implying some ED value and associated amounts of forgone-forestry, then this allocation of the areas can be represented as a point in a two-dimensional trade-off space. As an example, suppose that the forgone-forestry amounts for each area from Fig. 1a are those shown in Fig. 1b. These are the costs if the areas are protected rather than managed for forestry production. Suppose that the areas shown in bold define the current set of protected areas in this region and that all other areas will be logged. This set of areas has an ED value of 365; this is the relative amount of forgone biodiversity if only these areas are protected. Compared to other sets of 6 areas, this set spans the space well. The total forgone forestry associated with this set is 40 units. This land use allocation is represented as a point in Fig. 2.

This point can be used to calculate a measure of sustainability for the simple case where there are known standards or targets for the individual criteria. For example, in the AMOEBA system (ten Brink, 1991), a sustainability rating is indicated by calculating a distance of the actual point from the ideal or target point. Suppose that the trade-off space of Fig. 2 provides the ideal value for each

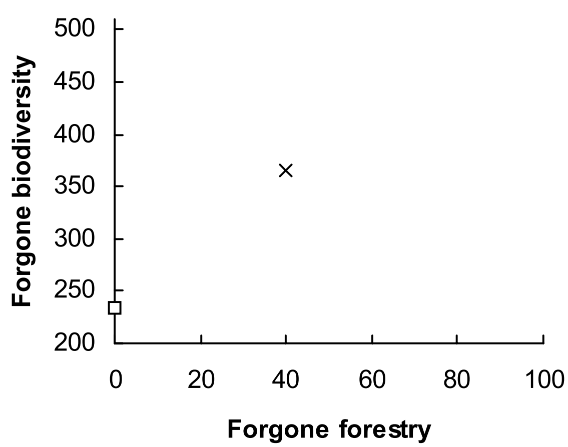

Figure 2. The total forgone biodiversity and total forgone forestry for the allocation shown in Fig. 1b are represented by the point shown as an x. The minimum possible forgone biodiversity (ED = 233 when all areas are protected) and minimum possible forgone forestry (0 when all areas are logged) is represented by the hollow square.

Box 1

Formulae for sustainability indices. w1 and w2 are weights.

S1 = 1 - .5 [w1(actual forgone-biodiversity r.s.v. - forgone-biodiversity target r.s.v.) + w2(actual forgone-forestry r.s.v. - forgone-forestry target r.s.v.)],

$$S2 = 1 - \frac{\text{w1(actual forgone-biodiversity - forgone-biodiversity target) + w2 (actual forgone-forestry - forgone-forestry target)}}{\text{w1 (range of possible forgone-biodiversity values) + w2 (range of possible forgone forestry values)}}$$

criterion (0 forgone forestry opportunity and some minimum value for ED). The sustainability rating is given by some function of the sum of the distances of the actual point from the target for each criterion (distance expressed as a proportion of the range of the criterion values). In this example, the range for forgone forestry is 103 (the sum of the forestry suitabilities for all areas, minus the minimum of 0) and the range for biodiversity is 1101 (maximum of 1334 minus minimum of 233). Biodiversity protection is (365-233)/1101 of the maximum-possible distance from the ideal value, and forgone forestry is 40/103 of the maximum-possible distance from its ideal. The current allocation therefore has a sustainability index value, *S1*, of:

$$1 - .5[132/1101 + 40/103] = 0.75$$

The two distance components have been averaged and this average subtracted from 1 so that a higher value of *S1* corresponds to higher sustainability.

One limitation of this approach is that the two criteria are arbitrarily weighted by their ranges. A better approach is to acknowledge that any overall measure must explicitly consider some relative weighting on the criteria. *S1* can be re-defined with weights and range-standardised values ("r.s.v."), as shown in Box 1. In practice, for this simple case of two criteria, both weights are not needed, and a weight w can be considered for the forgone-forestry amounts only.

In the formula for *S2*, the range standardisation is applied as a weighted sum of individual ranges, in contrast to the application of a range standardisation independently to each criterion as in AMOEBA and the *S1* measure[2]. The weighted sum standardisation maintains additivity of costs. Additivity requires that a change of k units in criterion 1 should have the same effect on the sustainability value as a change of w_2/w_1 times k units in criterion 2. For example, in Table 1, *S2* values are the same for a change (comparing scenario I with scenario II) in criterion 1 (with weight =1.0) of 2 units and an alternative change (comparing scenario I with scenario III) in criterion 2 (weight = .5) of 1 unit. In contrast the alternative index, *S1*, with individual-range standardisation (*S1* = 1 - the average of actual-minus-target values, after range standardisation), yields a different result for scenarios II and III.

S2 is similar to a weighted Manhattan metric (a measure that adds up absolute values of differences; Sokal and Sneath, 1973), with one important difference: the absolute value is

---

2

In general, for N criteria, $S2 = 1 - \{(\text{sum}_{(i=1 \text{ to } N)} [w_i(a_i - t_i)]) / (\text{sum}_{(i=1 \text{ to } N)} [w_i \times \text{range}_i])\}$ where a_i is the actual value for the ith criterion, t_i is the target value, w_i is the weight, and range_i is the range for the ith criterion.

Table 1.

Three regional scenarios, I, II, III. Criterion 1 has a weight of 1.0 and criterion 2 a weight of 0.5. The values shown are the forgone values for each criterion. Only for index *S2* is a change of 2 units in criterion 1 equivalent to a change of 4 units in criterion 2.

I	criterion	range	actual value	target value
	1	50	20	10
	2	500	200	100
S1 = .850				
S2 = .800				
II	criterion	range	actual value	target value
	1	50	22	10
	2	500	200	100
S1 = .830				
S2 = .793				
III	criterion	range	actual value	target value
	1	50	20	10
	2	500	204	100
S1 = .848				
S2 = .793				

not used. This means that any particular actual-minus-target value contribution for a given criterion could be negative. In contrast, the distance-contributions in AMOEBA for individual criteria will never be negative; a value falling below a standard level would contribute a positive distance (ten Brink, 1991). This restriction also exists for other general multicriteria measures that allow weights on the criteria. For example, Nijkamp (1979) considers weighted-distances to ideal points in a regional allocation context, but uses measures that are all variants of Minkowski metrics; these measures (of which the Manhattan metric is one special case) all effectively take the absolute value of differences between actual and target values.

It may seem odd to be concerned about allowing for the possibility of negative difference values for a given criterion, since the standard or target value represents some minimal amount of forgone opportunity that generally is not reached by actual values. However, the next section introduces an alternative to the simple targets or ideals based on individual standards for different criteria, replacing these with targets defined by optimal trade-off solutions. Such a trade-off target means that the actual value on any one criterion may well be lower (better) than that found in the best overall trade-off solution.

The absence of an absolute value also means that the terms in *S2* can be rearranged (Box 2). TNC stands for total net cost. In the examples presented here, TNC is the weighted sum of the forgone biodiversity and forestry values. The formula could be expressed alternatively using the complement, total net *benefit*, but it will be more convenient here to calculate cost values.

Box 2.

$$S2 = 1 - \frac{\text{(actual-forgone-biodiversity minus forgone-biodiversity target)} + w\,\text{(actual forgone-forestry minus forgone-forestry target)}}{\text{(range of possible forgone-biodiversity values)} + w\,\text{(range of possible forgone forestry values)}}$$

$$= 1 - \frac{[\text{(actual-forgone-biodiversity)} + w\,\text{(actual forgone-forestry)}] - [\text{(forgone-biodiversity target)} + w\,\text{(forgone-forestry target)}]}{\text{(range of possible forgone-biodiversity values)} + w\,\text{(range of possible forgone forestry values)}}$$

$$= 1 - \frac{\text{actual TNC} - \text{target TNC}}{\text{range of TNC values}}$$

Standards or targets and the capacity for trade-offs

The measures described above calculate a distance from the current state to a target, with weights on criteria taken into account. One difficulty in applying such a system of weighting is that there can be a wide range of plausible weights. This problem is confounded by a further difficulty: an ideal (or standard or target) value may not be easy to identify for many of the criteria. This is a key issue in considering regional biodiversity protection; one interpretation of sustainability argues that we should adopt a form of "safe minimum standard" (SMS) that allows *no* reduction in biodiversity (unless costs are intolerable; see discussion below). Pearce and Turner (1990), for example, argue that "maximising the net benefits of economic development, subject to maintaining the services and the quality of the stock of natural resources over time, is an essential criterion for sustainable development". Taken at face value, this suggests that in the regional context all areas would have to be protected (for example, all 17 areas in the scenario shown in Fig. 1) because each makes some contribution to biodiversity. The weakness of this "no-loss" argument, at least when it is interpreted literally at the regional level, where areas are to be allocated to biodiversity protection or to development, is that the standard is impractical.

An alternative way to view the SMS is that the decisions about allocation of areas to protection (e.g. to nature reserves) can be taken as only indirectly indicating possible "loss" of biodiversity. Different candidate sets of protected areas are compared, using a measure such as ED, only in terms of the expected relative degree to which regional biodiversity is protected. This can be interpreted as the degree to which we can be confidant that these is no loss of biodiversity. Thus, while the SMS might be accepted in principle, in practice the degree of protection of regional biodiversity is appropriately subject to trade-offs with other criteria (see Faith and Walker, in press b). This perspective is compatible with the argument that sustainability requires that total net benefits are to be maximised.

Turner (1990) discusses the SMS in the context of land allocations, suggesting that the "intolerable costs" must be defined via trade-offs with social opportunity costs. In his discussion, the allocation-SMS is seen as equivalent to an identified set of priority areas, to be protected unless costs are intolerable. It appears that such a set of areas is identified without regard to alternatives that might have lower cost. Faith and Walker (in press b) discuss examples in the context of land allocation, where a given allocation has an intolerable cost if some other allocation could provide greater net benefits over a wide range of weightings on the critical criteria. This contrasts with Turner's strategy in an important way; here, the priority areas are defined as part of the process of examining trade-offs.

Including biodiversity in regional trade-offs suggests that the formula given above for sustainability, *S2*, could be applied, using targets based on individual criteria as before. Depending on assigned criterion weights, different degrees of satisfaction of the criteria are achieved, corresponding to different forms of compromise. An alternative is to have the target instead indicate the trade-offs that are possible in the region, so that sustainability reflects the degree to which this capacity for actual trade-offs is achieved. Here, a substitute is needed for the a-priori standards or targets for individual criteria, that reflects instead a goal relating to the combination of different criteria.

It was argued above that weights have to be considered explicitly when exploring trade-offs; these weights also will play an important role in determining the substitute for an a priori ideal or target point. Each nominated weighting determines an ideal point in trade-off space, in the sense of identifying one particular allocation that would represent the best trade-offs, equivalent to best total-net benefits. Overall sustainability can be assessed over a range of weightings.

In considering trade-offs, the complementarity principle referred to above has important implications. The contribution of an area to the overall represented biodiversity may or may not exceed its value for a competing land use, depending on which other areas are taken to already be protected (for discussion see Faith and Walker, 1994; in press b). If the costs associated with protection of the different areas correspond, for example, to forgone forestry opportunity, then a trade-off

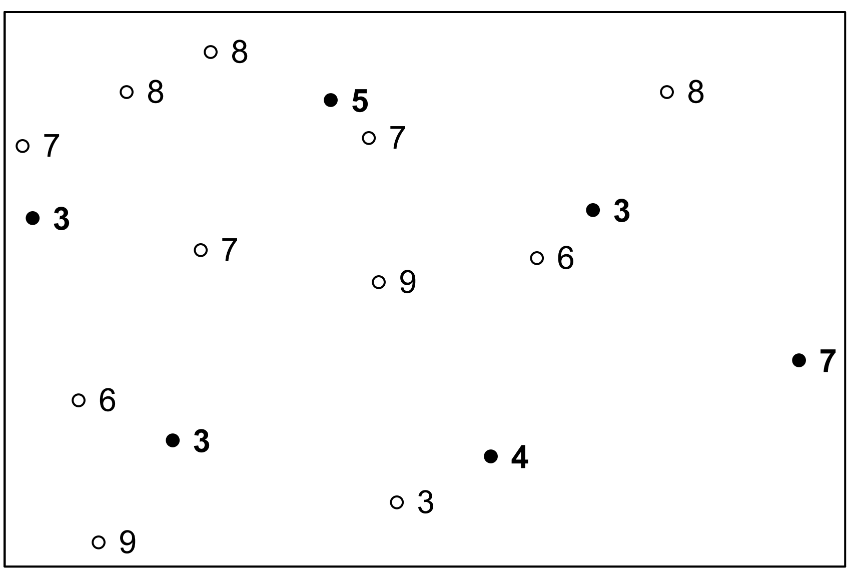

Figure 3. An alternative set of six protected areas, also having a forgone biodiversity of 365, but with lower associated cost or forgone forestry (25 units). If forestry is given a weight of 5, then this allocation of areas to protection, with the remainder allocated to forestry, is optimal in providing maximum possible total net benefit.

between forestry and biodiversity, maximising net benefits, can be accomplished by selecting only those areas for protection whose final contribution to biodiversity (in the examples presented here, the extent to which the area improves the ED value) exceeds its weighted cost (Faith and Walker, 1994; in press b).

Returning to the example of Fig. 1, an alternative set of 6 protected areas is shown in Fig. 3; this set also has an ED of 365, but the total forgone forestry is only 25 units. This example allocation was chosen such that, for a range of weights from 4 to 7 on forestry, the individual contribution of each of these areas (in the context of the complete set of protected areas) to lowering the ED value in each case exceeds the associated weighted forgone forestry. The total net cost, TNC (see formula above for *S2*), for this set of areas is minimised:

TNC = ED + w (total forgone forestry)
= 365 + (w x 25).

Suppose the weight is taken to be 5. TNC is then 500. Area 9, if removed from protection, would increase ED (increase forgone biodiversity) by 45 units, and correspond to avoidance of a weighted cost of only 5 x 5 = 25 weighted units. Consequently, the TNC would be greater (520). As an alternative to the protection of area 9, area 8 might have been chosen for protection; its contribution to lowering ED (43) would have exceeded its weighted forgone forestry (5 x 7 = 35). However, the resulting ED value (total forgone biodiversity) would be slightly higher than for area 8, with a weighted forgone forestry of 35 that is also greater. The TNC when area 8 is substituted for area 9 is 520 - 43 + 35 = 512.

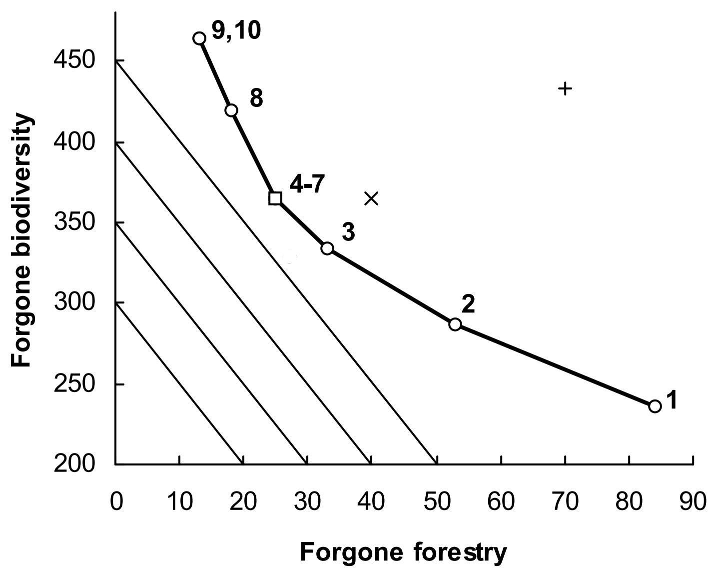

Figure 4 a) A trade-off or regional sustainability space for the 17 areas and the forestry opportunities shown in Fig. 1,3. Any given allocation of the areas to protection or forestry results in a total cost and total forgone biodiversity value, so that the allocation can be plotted as a point in this space. The allocation of 6 areas to protection under the costs scenario of Fig. 3 is shown as a hollow square. A given weighting on costs implies a set of equal-total-net-cost (TNC) contours in the space; the contours shown are for a weight of 5. The allocations having minimum TNC for different weightings (numbers next to points) is shown by the set of hollow points forming a trade-off curve in the space. The x represents the less-optimal allocation of six areas to protection shown in Fig. 1b and Fig. 2. The + represents the allocation of the first 12 areas to protection.

Given the protection of area 9, complementarity means that the similar area, 8, now has a smaller contribution to lowering ED when area 8 is protected (reducing ED by only 7.6). Therefore, TNC will not be reduced by additionally protecting area 8, given its cost of 35 (weighted) units.

Given these 6 protected areas, no other area is justified for protection (for the weighting of 5); any additional area would reduce forgone biodiversity by a number of units that is smaller than its weighted forgone forestry.

Naturally, if the weight given to forestry were greater, a different set of areas would satisfy the trade-offs constraint. For a greater weight, the number of protected areas would be expected to be smaller; for a smaller weight, the number of protected areas would be larger.

The forgone-biodiversity and forgone forestry values for the set of protected areas shown in Fig. 3 can be plotted in a trade-off space (Fig. 2), whose axes are costs (forgone forestry opportunity) and forgone biodiversity. The corresponding values for those allocations having minimal TNC for other nominated weights also can be plotted (Fig. 4a). The result is a trade-off curve. The lines in Fig. 4a show TNC equal-value contours when the weight is 5.

The allocation from Fig. 1b also can be plotted in this space (the x in Fig. 4a); this point is directly to the right of the optimal solution (Fig. 3) for weights 4-7; this reflects the equivalent forgone biodiversity but higher forgone forestry for the allocation of Fig. 1b. For comparison, a solution corresponding to

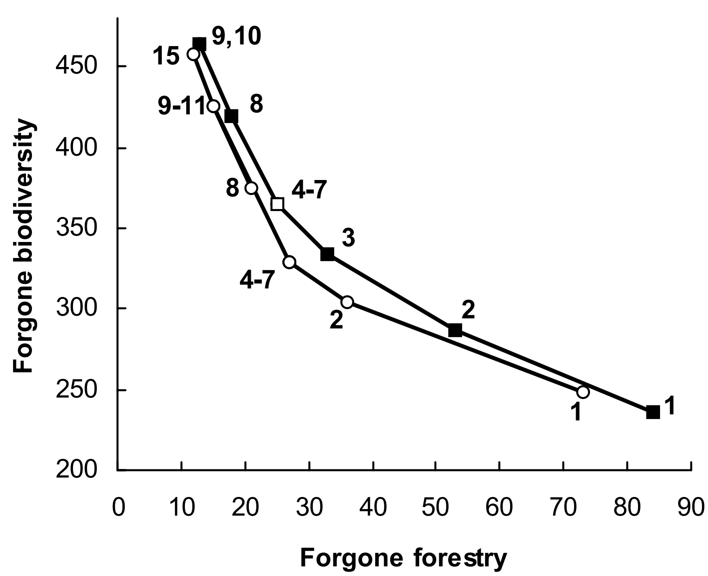

Figure 4 b) The same trade-off space as in a), showing an additional curve for the distribution of costs shown in Fig. 5a,b. The positions of all points represented by hollow symbols, including the hollow square representing the allocation of Fig. 3, are those for the modified costs of Fig. 5. The solid points are those allocations whose positions in trade-off space were calculated for the original costs of Figs. 1,3. Weights are shown next to points. Under the new distribution of costs among the 17 areas, the original allocation is no longer optimal; the lower curve shows that, for a weight of 5 for example, it now is possible to have lower forgone biodiversity for about the same total cost.

protection of the first 12 areas (and logging of the remainder) also is shown (Fig. 4a). The forgone biodiversity is much greater than that for the 6 protected areas, illustrating that it is not simply the total number of areas protected that is critical to the amount of biodiversity protected. Given that 12 areas are protected, the total cost also is much higher.

This example suggests that the solution of Fig. 3 may be given a high sustainability rating, while the solution of Fig. 1b and the solution corresponding to the set of 12 protected areas (with all others logged) both would be rated as less sustainable, with the latter solution having a particularly poor rating.

These assessments can be quantified using the formula for *S2* introduced earlier, but with one important difference: the ideal or target point will be the allocation representing the best-possible trade-off for the nominated weighting. A general formula for sustainability, "*S*", which includes *S2* as a special case, is given by:

$$S = 1 - \frac{\text{actual TNC} - \text{target TNC}}{\text{range of TNC values}}$$

The range standardisation in *S* optionally could be ignored, particularly for evaluations within a single region.

As an example, *S* will be applied to the scenarios of Figs. 1-4. Let the nominated weight be 5. For the allocation of Fig. 1b, the actual TNC was 365 + (5 x 40) = 565. The range of TNC values is 1101 + (5 x 103) = 1604. The target is the best-possible allocation for this weighting (Fig. 3), having a TNC of 500. Therefore *S* = 96%. For the allocation of the first 12 areas to protection (Fig. 4), the *S* value for the same weighting is only 82%.

This evaluation of regional allocations, based on the degree to which biodiversity and forestry are effectively traded-off, highlights the fact that sustainability will not be indicated well by using the criteria separately. To extend the example of Figs. 1-4, suppose we have another region in which the environmental space is the same, and the set of protected areas

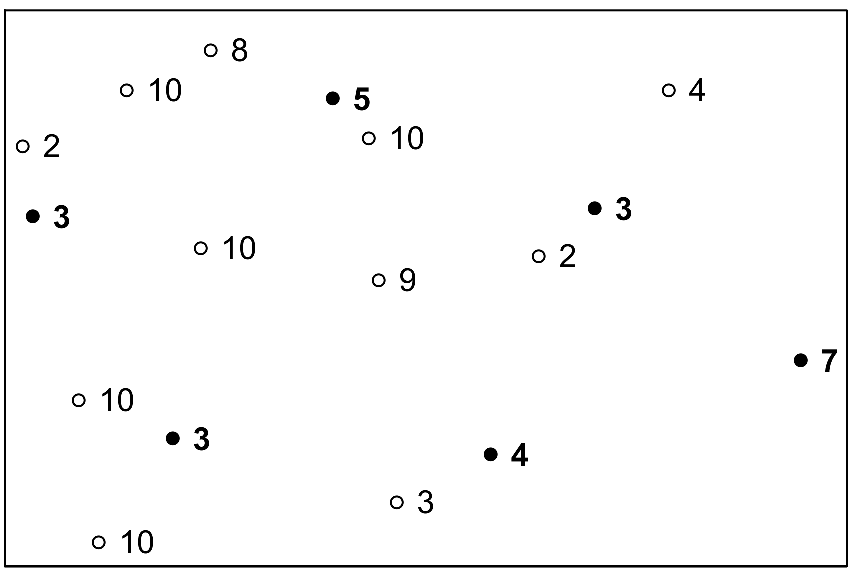

Figure 5a) Here the same six areas are protected with the same total cost s in Fig. 3; however, while the total forestry opportunity over the 17 area is unchanged, the distribution of this opportunity over the unprotected areas has changed. Under this scenario, even though total represented biodiversity and total forgone forestry is unchanged, this allocation is no longer optimal.

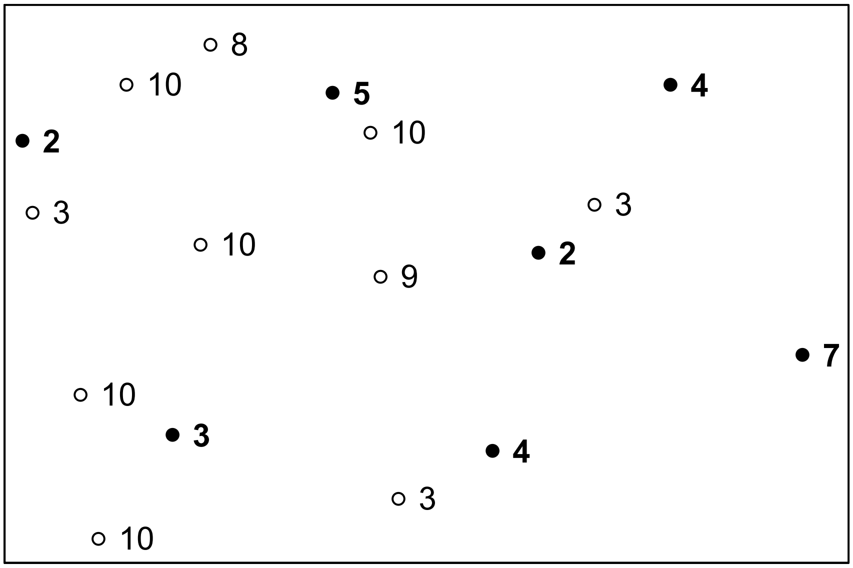

Figure 5b) For the weighting of 5, this new allocation of areas to protection would provide maximum total net benefit (minimum total net cost, TNC).

is the same, with the same forgone-forestry. Further, suppose that, while the total forgone forestry over all 17 areas is the same for the two regions, the **distribution** of these forgone-forestry amounts (among the non-protected areas) is different (Fig. 5a). This second region might appear to have the same sustainability; the amounts of biodiversity and forestry forgone (in absolute amounts or as a proportion of the maximum possible) are the same for the two regions. However, this new region differs from the first in the degree to which optimal trade-offs have been achieved, as revealed in the new trade-off curve (Fig. 4b). For weights of 4-7, this new region has a potential allocation with about the same total cost as for the first region, but with a significantly reduced forgone biodiversity. This solution is shown in Fig. 5b.

For this new region, the sustainability, *S*, will reflect the degree to which the actual allocation has achieved the trade-off that was potentially achievable, for the given weighting. This new trade-off curve now can be used to provide the ideal or target values for any given weighting.

As an example, for a weight of 5, the target allocation (Fig. 4b) is given by the hollow circle for that weight with values (27, 329.4). The initial allocation of six protected areas (Fig. 5a), shown as a hollow square in Fig. 4b, can be evaluated. That allocation had values (25, 364.8), resulting in a sustainability rating of:

$$S = 1 - \frac{(364.8 - 329.4) + 5(25 - 27)}{1101 + 5(103)}$$

$$= 1 - \frac{35.4 - 10}{1616}$$

$$= 98\%$$

Thus, the hollow-square allocation had a 100% sustainability at weight 5 for the first region (Figs. 3,4a) but 98% for the second region (Figs. 4b,5a). In contrast, the allocation of Fig. 5b would have a sustainability (for weight = 5) of 100%; this allocation corresponds to the best-possible allocation in minimising TNC (Fig. 4b).

Constraints and regional sustainability analysis

Another region now will be considered, having the same 17 areas and forgone-forestry values as in Fig. 5a,b but with constraints that areas 1 through 5 are to be protected, while areas 15 and 16 are to be logged (Figs. 1a,6a). For a given weighting, the DIVERSITY package (Walker and Faith, 1993, 1994) can be used to find the best trade-off allocations, taking these constraints into account. Again, a weight of 5 will be chosen for illustration. For this weight, the optimal trade-off solution is shown in Fig. 6b. The additional protection of areas 7, 9, and 15, with the remaining unconstrained areas logged, is optimal in minimising total net cost, TNC.

Fig. 7a shows a redrawing of the trade-off space from Fig. 4, with this new constrained analysis included. The lower curve from Fig. 4b has been redrawn here, as it again provides reference allocations for this region, in the absence of constraints. The hollow square represents the starting allocation, given the constraints (areas 1-5 are protected; all others therefore are taken to be logged). The curve with the solid squares then shows the optimal solutions available for different weights. In the figure, solutions for the same weight are connected by a line segment connecting the two trade-off curves. A consequence of the starting constraints is that, no matter what the weight, it is not possible to get very close to the trade-off curve found with no constraints.

The scenarios in Fig. 7 suggest alternative ways of viewing sustainability. The current allocation shown by the hollow square can be compared to the target points along the lower curve, or to the points along the constraint curve (Table 2). Alternatively, the best-possible trade-off for a given weight, under the constraints, could be compared to the corresponding value from the lower curve which assumed no constraints. All these alternatives depend on a choice of weight in order to determine the ideal and/or the "observed" allocations.

"Current sustainability" will refer to comparisons of whatever is defined as the current allocation to some standard - either a best-possible in the absence of any constraints or the best possible acknowledging some set of constraints. "Potential sustainability" differs from current sustainability in comparing the best that can be achieved under a set of

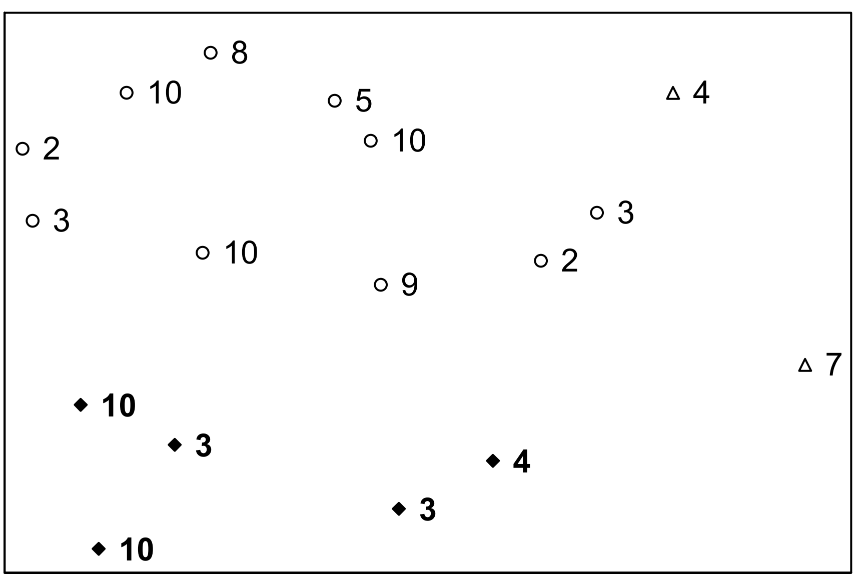

Figure 6.a) For the same costs as in Fig. 5a,b, areas 1 through 5 (solid diamonds) are constrained to be protected, and areas 15 and 16 (hollow triangles) are constrained to be logged.

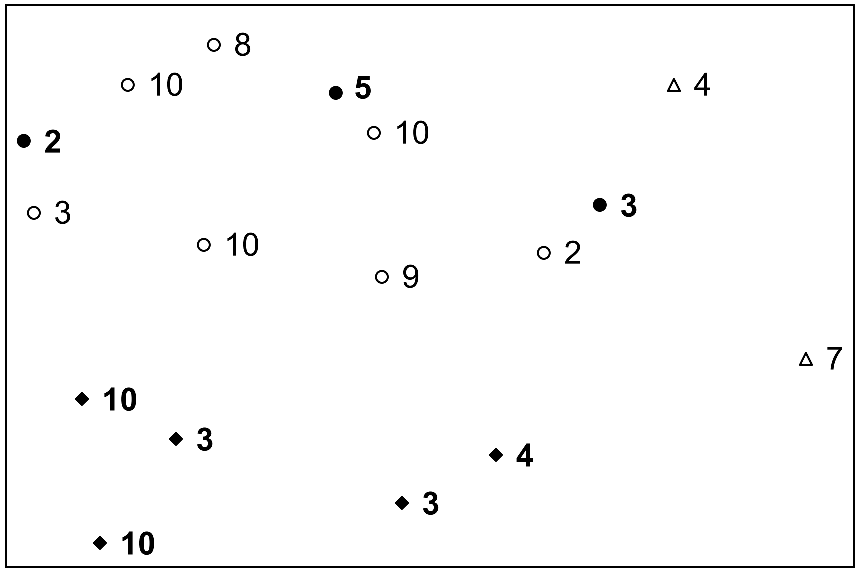

Figure 6b) For a weighting of 5, additional protection of areas 7, 9, and 15 minimises TNC, given the constraints.

Box 3.

Alternative calculations of regional sustainability, S, based on the trade-off space of Fig. 7a,b.

"current sustainability" = distance from current allocation to best-possible-with-no-constraints:
for w = 2, $S = 1 - (761 - 376) / (1101 + 2 \times 103) = 71\%$
for w = 15, $S = 1 - (1151 - 638) / (1101 + 15 \times 103) = 81\%$

"current constrained sustainability" = distance from actual current allocation to best-possible-with-given-constraints:
for w = 2, $S = 1 - (761 - 455) / (1101 + 2 \times 103) = 77\%$
for w = 15, $S = 1 - (1151 - 946) / (1101 + 15 \times 103) = 92\%$

"potential sustainability" = distance from best-possible-with-given-constraints to best-possible-with-no-constraints:
for w = 2, $S = 1 - (455 - 376) / (1101 + 15 \times 103) = 94\%$
for w = 15, $S = 1 - (946 - 638) / (1101 + 2 \times 103) = 88\%$

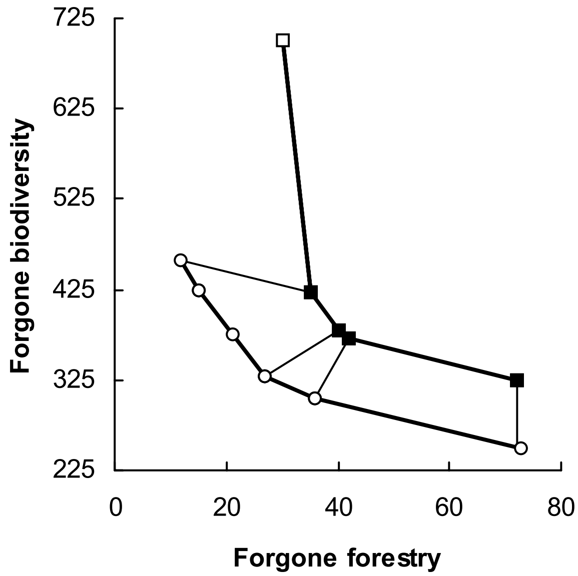

Figure 7a) A trade-off space in which the lower curve is identical to the lower trade-off curve in Fig. 4b, for weights ranging from 15 (at left) down to 2 (at right). The upper curve represents the corresponding trade-off curve for the same range of weights under the constraints shown in Fig. 6. Lines connecting the two curves link allocations for the same weighting. The hollow square represents the initial constrained allocation of areas 1-5 to protection and 15, 16 (plus other unprotected areas) to forestry.

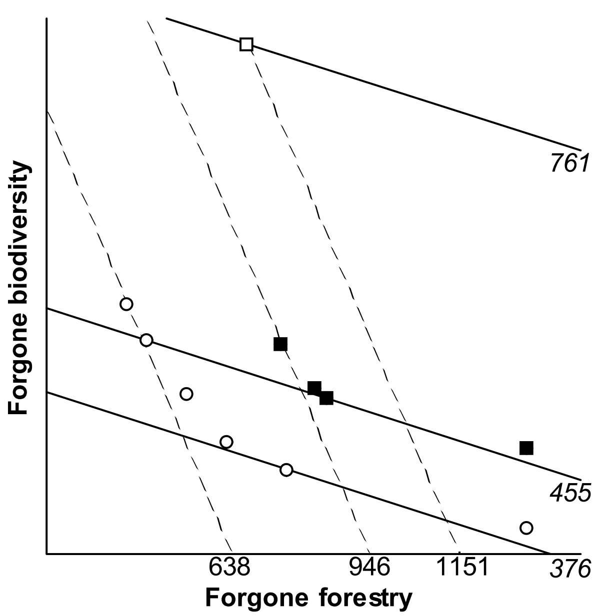

Figure 7b) A redrawing of the trade-off space showing those TNC contours crossing through optimal or non-optimal allocations, for weights of 15 (dashed lines) and 2 (solid lines). Axis values (not shown) are the same as in a). TNC values are shown at the ends of each contour.

recognised constraints to the best-possible in the absence of these constraints.

Fig. 7b shows a redrawing of the points along the two trade-off curves from Fig. 7a, with equal-TNC contour lines shown for weights of 2 (solid lines) and 15 (dashed lines). These values are used for the calculations of regional sustainability in Table 2. The current sustainability falls between 71% and 81% over the range of weights examined. Over this same range of weights, the potential sustainability, given these constraints, ranges from 88% to 94%. These results illustrate how information about the status of the region's sustainability is gained even without assuming a single "correct" weight.

It is also revealing to compare the potential sustainability for high versus lower weights. At the high weight of 15, the current allocation has an *S* value of 81%, as compared to a lower value of 71% for a weight of 2. However, for the high weight there is relatively little scope for improvement; the potential sustainability value is only 88% (Fig. 7a). In contrast, while the current sustainability value, *S*, is only 71% for the weight of 2, the best possible allocation for this weight implies a potential sustainability value of 94%. Thus, while the current allocation and constraints may appear to imply low sustainability when forestry is weighted low, potential sustainability is in fact highest when the weighting is low. The reason is that the constraints, in implying the protection of 5 areas, foreclose forestry opportunity much more than the opportunity for biodiversity-protection.

Fig. 8 presents an example with another form of constraint. Here, the same costs from Fig. 5a,b apply and now all even-numbered areas (Fig. 1a) are assumed as a form of constraint to be cleared land, with no contribution by these areas possible (initially) either to forestry or to biodiversity protection. The ideal (ignoring the constraints regarding cleared land) trade-off curve (hollow circles), along with two other curves, are shown in Fig. 8. The five points along all three curves in Fig. 8 correspond to weights 1, 2, 5, 10, 15. The upper-most curve (small solid squares, with *S* values indicated next to each point) represents the curve for potential sustainability under the constraint of no biodiversity or forestry benefits from the cleared land. As in the previous example, each solution along this curve represents an allocation such that, for a given weight, any available area is assigned to biodiversity protection if its final biodiversity (ED) contribution exceeds the weighted cost. Relative to the ideal curve, in the upper curve each solution for a given weight has both higher forgone biodiversity and higher forgone forestry because of the opportunities lost to land-clearance. For high forestry weight, the sustainability is particularly low, because nearly 60 units of forestry opportunity cannot be realised, even in the absence of demand for biodiversity protection. In contrast, for low forestry weight, the potential sustainability is nearly 90%; even while many areas are not available for forestry or biodiversity protection, the absence of forestry pressure allows substitute areas (odd-numbered areas in Fig. 1a) to be found that still span the environmental space reasonably well.

The middle curve represents a slight alteration in the constraints in which some portion of the cleared land (even-numbered areas) is assumed to be used for forestry plantations, with a total reduction in forgone forestry of 20 additional units (independent of allocations of the odd-numbered areas to biodiversity or forestry). This forestry benefit from plantations might be a total estimated over a 50 year period, taking into account some delay before any harvesting is possible.

The resulting new trade-off curve is identical to the first one but is shifted to the left by 20 units. Note that this results in a large "pay-off" in increased potential sustainability only for the case where forestry is weighted highly. For a low weight on forestry, only the forgone biodiversity really matters, and for the two constraint scenarios, the solution-point corresponding to a low weight of 1 (the lowest point on each curve) will intersect approximately the same TNC contour.

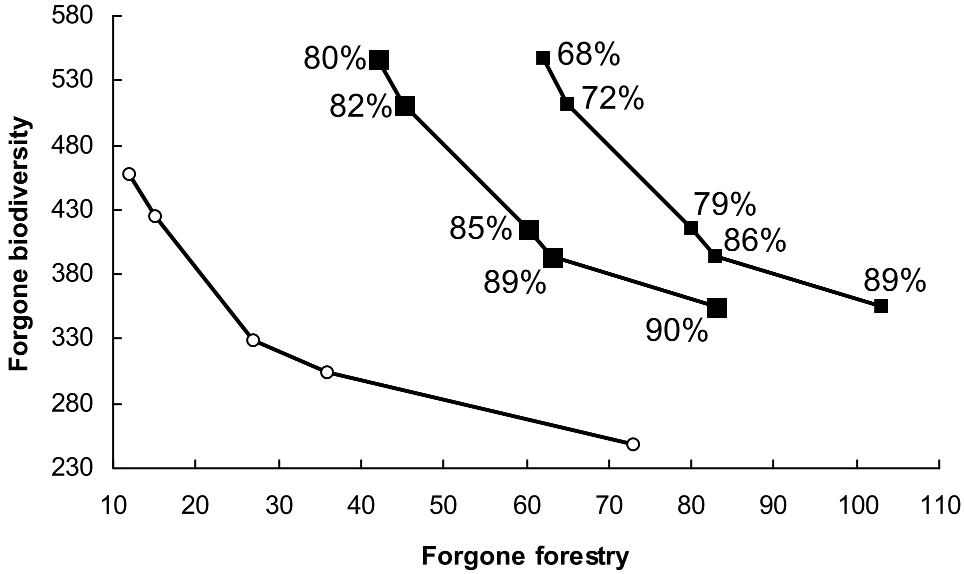

Figure 8. The lower trade-off curve (with open circles) is that of Fig. 4b (lower curve) and Fig. 7 (lower curve), for the costs from Fig. 5a,b. The two upper trade-off curves (solid squares) are those for the scenarios where all even-numbered areas (Fig. 1a) are taken to be cleared. The five points along each curve correspond to weights 1, 2, 5, 10, 15. The upper-most curve (small solid squares) represents the case where the cleared areas do not contribute to either biodiversity protection or forestry. The middle curve (large solid squares) represents the case where some of these cleared areas are used for plantations, with a regional contribution of 20 forestry opportunity units. The percentages along each curve represent potential sustainability values, where the ideal or target for each weight is defined by a point along the lower (open circle) curve. When forestry is given a low weight of 1, there is little difference in regional sustainability between the plantation and non-plantation cases; both scenarios are dominated by a shortfall in biodiversity protection. However, when forestry is given high weight of 15, the plantation case has a much better sustainability rating, even though the two scenarios again do not differ in forgone biodiversity.

Partial degrees of protection and regional sustainability analysis

The simple scenarios considered above viewed protection of biodiversity in an area as all-or-nothing; an area either contributed to the total representation of (protected) regional biodiversity or, if it was logged, was assumed to contribute nothing. This simplistic view ignores an important element of regional sustainability. This is the contribution that may be made at the level of individual areas, when (in our examples) both forestry and biodiversity opportunities are realised to some degree by a particular management regime. An example is presented here in which within-area sustainability contributions are taken into account in assessing overall regional sustainability.

An area can be assigned a partial degree of biodiversity protection (e.g. 50%) when that area has been allocated to some management regime that justifies a conclusion that (in the present context) there is some nominated level of biodiversity protection and forestry production. For an environmental space as in Fig. 1a, the ED measure can be adjusted to take into account such partial-protection values and capture the change in the overall expected forgone biodiversity given by any set of areas that includes such partial protection assignments (Faith and Walker, in press c).

As an example, Fig. 9a shows the 17 areas and the current optimal allocation from Fig. 5b for weights in the range 4-7. The distribution of forestry opportunities from Fig. 5b is shown, with the exception of two of the areas (1 and 8 marked with an "x"). For each of these areas, the cost of protection (forgone forestry opportunity) remains as 10 units under full biodiversity-protection, but alternatively is assumed to be only 1 unit if the area is logged "sympathetically" such that 50% of the biodiversity of the area is expected to be protected.

The optimal allocation now can be re-evaluated for various weights on forestry. For a weight of 7, the new solution is shown in Fig. 9b. Area 9 (Fig. 1a), which is environmentally similar to area 8, is now logged rather than protected, saving a forgone-forestry cost of 5 units. Area 8 is now sympathetically logged at

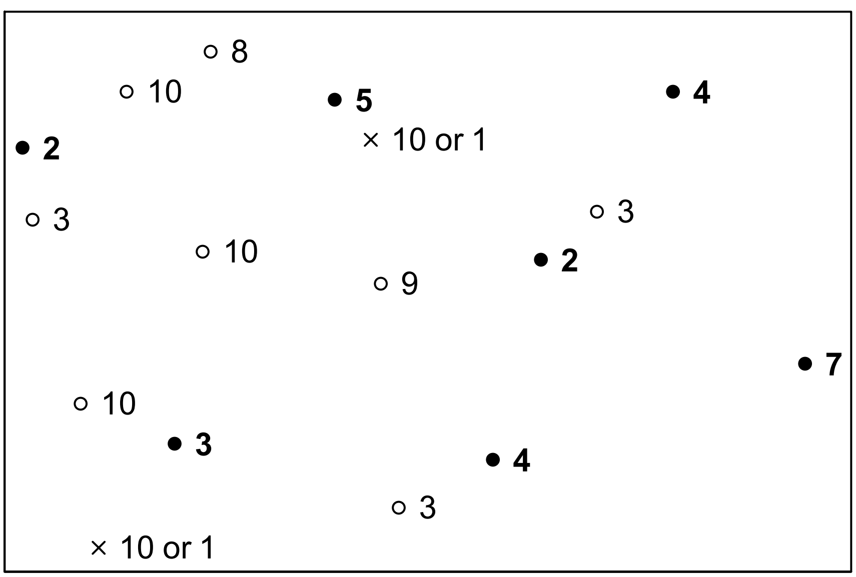

Figure 9a) the optimal allocation, for weights 4-7, from Fig. 5b is redrawn with modified costs for areas 1 and 8 (points now marked with an x). In each case, the cost of protection remains as 10 under full protection, but alternatively is only 1 unit if the area is logged sympathetically such that 50% of the biodiversity of the area is expected to be protected.

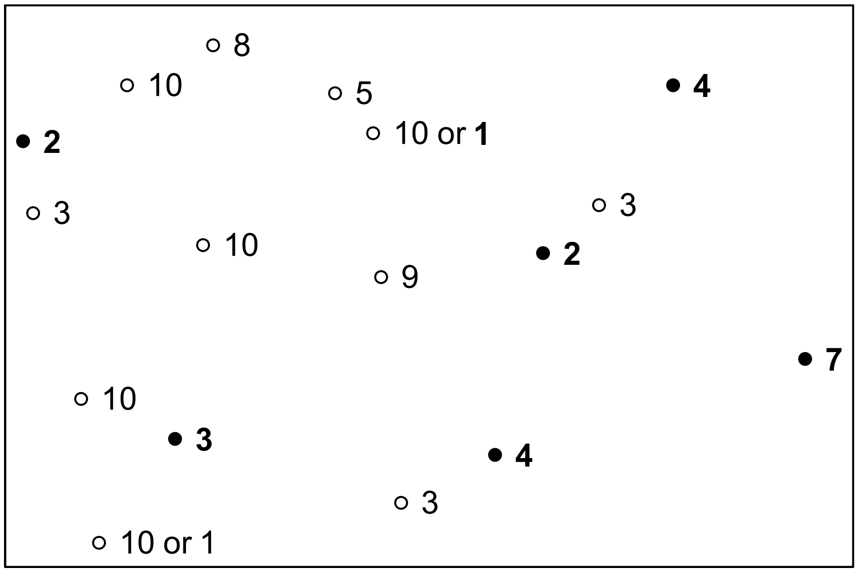

Figure 9b) For a weight of 7, it is now optimal to log area 9 (previously protected in the scenario of Fig. 5b), avoiding the previous cost of 5 units. It is also now optimal to sympathetically log (previously fully-logged) area 8 (with a resultant cost of 1 unit and with 50 % protection). The total regional cost is now reduced from 27 units (for the solution for weight = 4-7 in Fig. 5b) to 23 units, more than compensating (at this weight of 7) for a small increase in forgone biodiversity (see Fig. 10). For this weight it remains optimal to protect area 2 (bold point, lower left) and fully log areas 1 and 3 (the other two areas at the lower left). Area 2 fills such a large gap that the alternative of a 50% protection level on 1 would not produce the same net benefit; given the protection of 2, a 50% contribution from area 1 is then not greater than its associated cost of 1 unit.

a 50% protection level and with a cost of only 1 unit. The total cost is now reduced from 27 units (for the solution for weights 4-7, Fig. 5b) to 23 units, more than compensating (at this weight of 7) for a small increase in forgone biodiversity. This solution is represented by one of the solid circles along the lower trade-off curve in Fig. 10. Further, for this weight, it remains optimal to protect area 2 (bold point, lower left in Fig. 6b) and fully log areas 1 and 3. Area 2 fills such a large gap that the alternative of a 50% protection level on area 1 would not produce a comparable net benefit. Given the protection of area 2, the option of sensitive forestry of area 1 is not attractive, as a 50% contribution from this area is then not greater than its associated cost of 1 unit.

Under the scenario shown in Fig. 9 where partial protection is available for areas 1 and/or 8, application of a range of weights results in a new trade-off curve, as shown in the lower curve with solid circles (Fig. 10). For all weights, TNC is now reduced relative to the previous optimum value found under the scenario where no partial protection was possible. Over this range of weights, one or both of areas 1 and 8 are chosen for sympathetic logging with partial protection.

Partial protection, equivalent to a form of "sustainability" within individual areas, therefore has a measurable pay-off at the regional level. Note that prior to the consideration of the potential of these areas for sympathetic management, the trade-off curve represented by the hollow circles corresponded to solutions with 100% sustainability (*S* values of 100%). If this curve was retained as the standard, the new values along the lower curve would yield *S* values greater than 100%. In practice, the potential for partial protection in fact should define a new standard, so that the points along the lower curve now would have 100% values, while the previous optimal solutions then would be assessed as having high sustainability, but not quite 100%. Thus, the potential for sustainability contributions within areas has boosted the region's capacity for trading off conservation and forestry, with the result that regional allocations ignoring the potential for partial protection are not as sustainable.

This example also illustrates that the potential of an area to be sympathetically managed, and so appear to provide a level of sustainability within the area, does not automatically imply that such management will increase overall regional sustainability. Here, area 1, over a range of weights, actually made a greater contribution to overall sustainability when fully logged. Thus, the attractiveness of sustainable practices in individual areas ultimately must be evaluated at the regional level.

Incentives and regional sustainability analysis

The examples of regional sustainability analysis presented above illustrate how a region's performance can be compared to the trade-offs achievable in the region, with or without consideration of various constraints. The constraints discussed above played either of two roles: "accepted" constraints modified the perceived capacity of the region for trade-offs, while "unacceptable" constraints represented barriers to be overcome if the capacity for trade-offs was to be realised. The presence of such barriers highlights the potential gap between the initial **identification** of an optimal balance between biodiversity conservation and production in a region, and

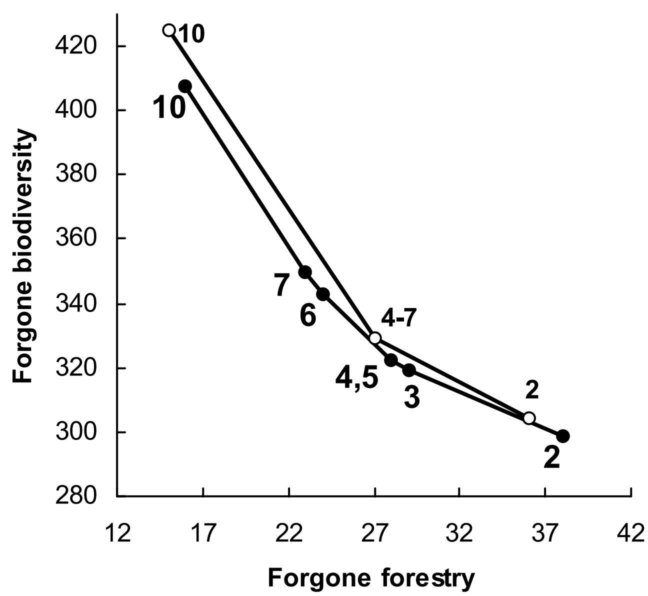

Figure 10. Under the scenario of Fig. 9, where partial protection is available for areas 1 and/or 8, a new trade-off curve results as shown in the lower curve with solid circles. Weights are shown next to points. For a weight of 7, TNC is now reduced relative to the previous optimum value under the scenario where no partial protection was possible. Over a range of weights one or both of areas 1 and 8 are chosen for sympathetic logging with partial biodiversity protection

the **process** of actually moving towards that goal. Policies and actions are needed to overcome such constraints to progress towards greater sustainability. Within the framework presented here, incentives will be an option at least for one particular form of constraint - arising when one extreme among the key criteria dominates the actual process of decision-making, so that the trade-offs would not be achieved. This may occur, for example, when the forestry sector has some control over the allocation process, resulting in allocations equivalent to having had a very high weight on forestry.

A general definition of an *incentive*, in the context of regional sustainability analysis, is any policy or action which alters the suitabilities (criterion values) of particular land-uses/management regimes in one or more areas, such that some desired more-sustainable solution is achievable, even under the decision-maker's extreme weighting. The criteria here can be any that form the trade-off space, as the incentives are intended to promote greater sustainability in the sense of trade-offs among all criteria (in contrast to the purely conservation-based incentives described in McNeely, 1988).

The scenario from Fig. 7 provides a simple example. A regional shift from the initial state (hollow square) to a solution along the constrained trade-off curve requires some increase in the total forgone forestry in order to achieve the potential for a dramatic increase in biodiversity protection. This shift may be promoted by an incentive (or "disincentive") in the form of a tax (or removal of existing subsidies) on those specific areas needed for protection, such that the net forestry-related gain for those areas is now close to zero. Such incentives need not be strictly "economic incentives" in the sense of McNeely (1988), and might be more generally referred to as "trade-off incentives". For example, any promotion of sensitive forestry that protects a large portion of the biodiversity of an area might encourage a conservation decision-maker to permit forestry in those areas, with a resultant gain in regional sustainability.

Regional sustainability analysis should provide a basis for evaluating possible incentives; as in the partial-protection example above, a particular sensitive management option may or may not promote regional sustainability, depending on the regional context.

DISCUSSION

Other multi-criteria methods

The examples presented here used two criteria, biodiversity and forestry opportunity, but the basic definition of sustainability is more generally applicable. A large number of different criteria, judged to be important by society in the given context, may be included. Over a range of weights on these criteria, the contrast between a current allocation and the ideal provides the basis for evaluating sustainability.

Applications of regional sustainability analysis that include a biodiversity criterion remain a notable special case of the framework developed here. Biodiversity assessment is only incorporated effectively into regional sustainability analysis when the biodiversity score for an area properly depends here on the set of other areas that are protected. This represents an important departure from other "scoring" approaches, sometimes combined with multi-criteria assessment, that only account for the biodiversity of individual areas (for review of such scoring approaches, see Spellerberg, 1992).

The assessment of regional sustainability therefore depends on this proper assessment of the relative biodiversity of sets of areas, together with the use of an effective distance measure that takes relative weights into account in comparing allocations. The key element of regional sustainability analysis is the calculation of the distance between a given allocation and the best-possible trade-off allocation. The examples given above illustrate how this differs from any approach that simply calculates the distance to a standard that is based on individual criteria.

Spellerberg has emphasised the limitations of weighting of different criteria in multicriteria analyses, and Petry (1990) has noted that weighted sums (as in the formula for S used here) are potentially misleading unless a wide range of values are examined. This problem also may be reduced when other constraints limit the range of possible solutions. For example, in Fig. 8 for the no-plantations scenario, if there is a constraint that forgone forestry must not be greater than 70 units, then the potential sustainability is constrained to be approximately 68 -72%. Further, the advantages from plantations, contributing 20 units regionally, is also now more clearly defined. The potential sustainability could be as high as 89% under this constraint of total costs less than 70 units (Fig. 8). Thus, a combination of sensitivity analysis and consideration of other constraints can yield useful insights, even if the weights are never fixed.

Sensitivity analysis applied to a range of weights also assists in identifying those solutions that are best-possible over the range of possible weights. For example, if two different solutions are offered, sensitivity analysis determines the range of weights where one solution will be superior to the other. In Fig. 4, in comparing the hollow square solution to the hollow circle solution for weights 4-7, it is apparent that the latter solution is better unless the weight given to forestry is very high.

As another example, consider the scenario where there were no constraints and the lower curve in Fig. 7 is achievable in principle. Suppose that there is a choice between the allocation represented by open circle at the lower right of the curve (73, 249) versus the allocation represented by the middle square (40, 380). For the circle, the S value (current sustainability) ranges from 99% (for a weight of 2) to a low 73% for a weight of 15. For the square, the range is 87% to 94%; on average it provides a higher sustainability rating. Thus, in this case the point that is not on the ideal curve has the better current sustainability, on average, over a range of weights.

This perspective contrasts with the recommendations of Munasinghe (1993; see also Hitchens et al.(1994)), who suggests a strategy that first attempts to find a solution on a trade-off curve, and then refines this by finding a preferred location on the curve. Nijkamp (1979) and Munda et al. (1994) also use the set of points along a trade-off curve in land-use assessments, but the points along the curve are not considered ideal points but rather possible solutions. The best of these points will be the one closest in some sense to an ideal point found elsewhere in the space. This approach is similar to those that define the ideal based on individual standards or targets for the different criteria.

Because the trade-off curve in the framework developed here forms the set of ideal points, the definition of the trade-off curve is slightly different from those applications where the curve forms potential solutions. The definition of the trade-off curve here also departs from the usual definition based on Pareto-optimality, used by Munasinghe (1993), Nijkamp (1979), Hitchens et al. 1978) and others. A Pareto-optimal point is one that occupies a position in the trade-off space such that there are no other solution-points in that region of the space where all the criteria have as-good or better values. Such a point is "non-dominated". In contrast, the trade-off curve used here, as the source of ideal points for calculating sustainability, is the set of all solution-points that are optimal (have minimum TNC) for some weighting[3]. This difference is highlighted by the example of Fig. 4b. The hollow-square solution is a non-dominated point for the set of costs that produced the lower trade-off curve; however, there is no defined weighting that would imply that the hollow-square solution has optimal net benefit (minimum TNC). Therefore, this solution is excluded from the trade-off curve defined here, which may be described as the set of all "weight-dependent-Pareto-optimal" points. -

[3] In the examples presented here, and for all analyses using the DIVERSITY package, the weight functions are strict linear combinations of the criteria; however, more generally the weight functions may be curved through the space.

These points would correspond to the set of all preferred alternatives in the sense of Lutz and Munasinghe (1994): a preferred alternative for a given weighting system is the one along the Pareto-optimum curve intersecting the highest equi-preference curve.

Cost-benefit analysis and regional sustainability

All of the methods described above, as variants of multi-criteria analyses, contrast with a cost-benefit analysis approach where different criteria are placed on the same economic cost-benefits scale (e.g. using contingent valuation). A cost-benefit approach effectively would define a single weighting of the biodiversity and forestry criteria used in the examples in this paper. This results in a simple trade-off space as illustrated, for example, in Fig. 3.2 of Pearce and Turner (1990), and redrawn below (Fig. 11a). A general measure of "natural capital" is used (see discussion above), which may be equated with biodiversity for comparisons with the approach developed here. Environmental "change" (Pearce and Turner, 1990) results in an increase or decrease of natural capital and associated costs. Both the costs, C, and the natural capital stock are placed on the same economic scale; for the latter, the curve designating benefits, B, translates a given amount of natural capital into cost-equivalents on the economic scale (Fig. 11a). Given this common scale, net benefits are maximised for that amount of natural capital stock which corresponds to the greatest excess of benefits over costs, $B - C$ (Fig. 11a).

The possibility of a recommended loss of natural capital stock as a result of such an analysis may seem surprising, given the argument noted above for a constraint of no-change in natural capital. Pearce and Turner (1990) indeed have argued that risk and uncertainty are so marked that reductions in natural capital should not be permitted, in spite of the possible implications of a such economic analysis. It was noted earlier in this paper that one resolution of this "no-loss" problem, at least in the context of regional biodiversity, is that

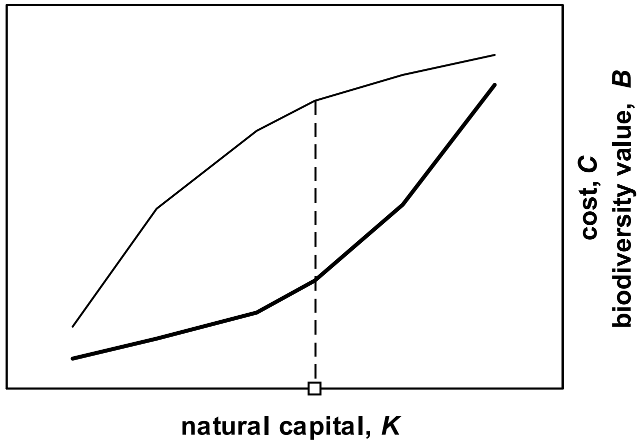

Figure 11 a) a redrawing of a cost-benefits space from Pearce and Turner (1990; fig. 3.2). Bold line represents increase in cost, *C*, as natural capital, *K*, increases. Lighter line represents the increase in benefits, *B*, arising from increase in natural capital. In the present context, natural capital could be equated with some measure of biodiversity, so that *B* is interpreted as the economic value of the amount of biodiversity protected. Because *B* and *C* are in the same economic units, they are represented along the same axis. The vertical dashed line is at that point (hollow square) along the natural capital dimension that maximises the difference between the benefits of that amount of natural capital and its associated cost. Note that the costs curve here can be compared to the trade-off curves from previous figures if the plot is rotated 90 degrees so that cost is along the horizontal axis and increases to the right, while forgone natural capital increases upward along the vertical axis. For further information, see text.

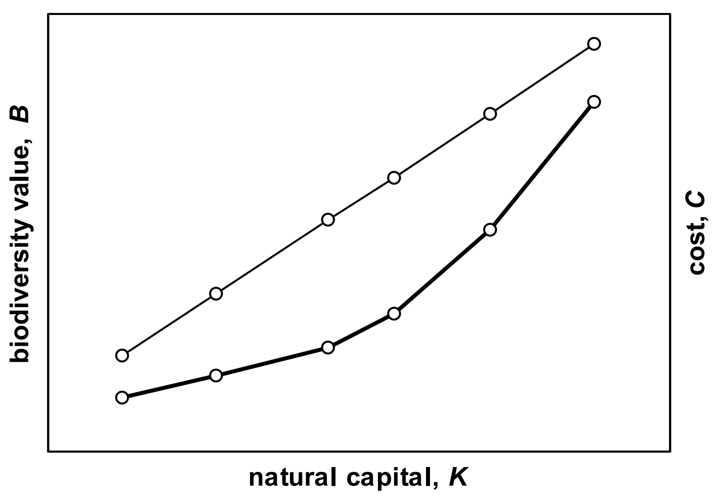

Figure 11 b) a modified version of a) compatible with a multicriteria-analysis perspective, and using the actual trade-off curve from Fig. 4 (rotated 90 degrees). Here *B* (light curve) and *C* (bold curve) are not on the same scale. The light curve is a straight line whose slope reflects only the relative weight given to *B* versus *C*, and is parallel with the equal-net-benefit lines for a given weight illustrated in Fig. 4.

alternative regional allocations are viewed as representing only different degrees of protection to overall regional biodiversity. This is an open-ended goal; consequently some trade-offs must be considered, given the absence of any other well-defined stopping rule for protection.

In a regional context, this perspective makes the prospect for deriving a common economic scale (as reflected in the curve for *B* in Fig. 11a) even more remote. In fact, the uncertainty about the values of biodiversity must mean that the relative weightings also must be uncertain. Therefore it is useful to explore how the cost-benefit space of Fig. 11a must be altered in order to accommodate such uncertainty.

In order to reflect regional biodiversity, the relationship between cost and natural capital of Fig. 11a first must be modified. A solution in the cost-benefit space (Fig. 11a) is always represented by two points, one along each of the curves, at the same natural capital value. A given natural capital value always corresponds in Fig.11a to the same given cost value. However, when natural capital is equated with biodiversity and a regional context is considered, this framework must be modified so that the same natural capital value (biodiversity value) may imply different cost values, depending on which particular areas in the region are allocated to protection so as to provide this biodiversity value (for example, contrast the solutions of Fig. 1 and 3). Therefore regional sustainability analysis requires replacing the simple cost curve of Fig. 11a with a trade-off curve, given by the optimum trade-off between cost and biodiversity for a given weighting.

As an example, the trade-off space from Fig. 4 can be redrawn (Fig. 11b) using the same axes as in Fig. 11a. The lower curve (Fig. 11b) corresponds to the lower trade-off curve of Fig. 4b. In Fig. 11b, the curve reflecting the relationship of natural capital to benefits, *B*, forms a straight line, because the biodiversity "value" and the surrogate measure for biodiversity are in the same units. The slope of this line varies with different weightings of biodiversity relative to costs. This line would form one of the equal net-benefit contours as in Fig. 4. Lines parallel to this one but closer to the lower right hand corner (Fib. 11b) would represent greater net benefit because points along these would have high biodiversity and low cost.

Given this modified version of the cost-benefit space, an optimum value for natural capital again is found by identifying the value that implies the greatest difference between *B* and *C*. This is equivalent to finding that point along the trade-off curve (lower curve) that intersects a net-benefit line closest to the right hand corner.

In conclusion, the standard cost-benefits approach as illustrated in Fig. 11a and in Pearce and Turner (1990) is transformed into the approach described in this paper (Fig. 11b) by taking the following into account:

1) Because there is no single weighting that transforms biodiversity into equivalent cost-units, the *B* curve becomes a set of equal-net-benefit contours dependent on a nominated weight.

2) Because the cost for a given amount of biodiversity depends on the particular areas chosen for protection, the curve, *C*, relating natural capital to cost is based on trade-offs.

3) While the set of possible solutions in Fig.11a falls along the curve, *C*, in Fig. 11b we have a continuous space of possible solutions, with the trade-off curve, *C*, defining those representing best trade-offs.

SUMMARY

One notion of sustainability links it to development that can go on without restricting options for future generations, for example derived from the environment. This sometimes leads to the impractical constraint of no loss of natural capital (e.g. no loss of biodiversity). A more pragmatic view of sustainability focuses on a requirement for effective trade-offs among competing demands, with the goal of maximising net benefits for society. The degree to which net benefits have been maximised in a given region is not indicated well by comparing the current situation, as represented by a list of parameters, to a corresponding list of standards; the capacity for trade-offs and the trade-offs actually achieved are not taken into account. Rather than define regional sustainability in terms of a list of indicators, it is defined here as the degree to which a region has achieved its own particular capacity for trade-offs.

The examples in this paper use biodiversity protection and forestry as an example of two partly-conflicting demands on land allocation in a region. The examples illustrate how useful information can be gained via sensitivity analysis, without nominating exact relative weightings of the criteria. The examples illustrate how

1) constraints on the set of areas given as already-protected versus already-forested dramatically affect current and potential regional sustainability,

2) land clearance reduces regional sustainability, but is partly compensated for by forest plantations on cleared land, and

3) a “sympathetic” form of forestry management of individual areas, providing partial biodiversity protection at a small cost, in some cases improves the overall sustainability rating of the region.

REFERENCES

Barbier, E. B. (1989) Economics, natural-resource scarcity and development: conventional and alternative views. Earthscan, London.

Birch, L. C. (1993) Regaining compassion for humanity and nature. NS.W. Press.

Bishop, R. C. (1993) Economic efficiency, sustainability, and biodiversity. *Ambio* 22: 69-73.

Brink, B. ten (1991) The AMOEBA approach as a useful tool for establishing sustainable development? in (eds. O. Kuik and H. Verbruggen) In search of indicators of sustainable development. Kluwer Academic,Dordrecht, pp71-87.

Ecologically sustainable development working groups (1991) Final report: ecologically sustainable forest use. AGPS, Canberra.

Faith, D. P. and Walker, P. A. (1994) *DIVERSITY: a software package for sampling phylogenetic and environmental diversity. Reference and user's guide. v. 1.0.* CSIRO Division of Wildlife and Ecology. Canberra.

Faith, D. P. and Walker, P. A. (1994) *DIVERSITY: a software package for sampling phylogenetic and environmental diversity. Reference and user's guide. v. 2.1.* CSIRO Division of Wildlife and Ecology. Canberra.

Faith, D. P., Walker, P. A., Ive, J. R. and Belbin, L. (1994) Integrating conservation and forestry production: effective trade-offs between biodiversity and production in regional land-use assessment. In: *Proceedings of International Forest Biodiversity Conference: Conserving biological diversity in temperate forest ecosystems - towards sustainable management*. A.N.U. Press.

Faith, D. P. and Walker, P. A. (in press a) Environmental diversity: on the best-possible use of surrogate data for assessing the relative biodiversity of sets of areas. *Biodiversity and Conservation*

Faith, D. P. and Walker, P. A. (in press b) Integrating conservation and development: effective trade-offs between biodiversity and cost in the selection of protected areas. *Biodiversity and Conservation*

Faith, D. P. and Walker, P. A. (in press c) Integrating conservation and development: incorporating vulnerability into biodiversity-assessment of areas. *Biodiversity and Conservation*

Faith, D. P. and Walker, P. A. (submitted) Hotspots and fire-stations: on the use of biotic and environmental data to estimate the relative biodiversity of different sets of areas. *Biodiversity Letters*.

Hitchens, M. T., Thampapillai, D. J. and Sinden, J. A. (1978) The opportunity cost criterion for land allocation. Review of Marketing and Agricultural Economics. 46:275-293.

IUCN (1980) World conservation strategy: living resource conservation for sustainable development.

IUCN (1991) Caring for the earth: a strategy for sustainable living. IUCN UNEP WWF, Gland, Switzerland.

Ive, J. and Cocks, K. D. (1988) LUPIS: A decision-support system for land planners and managers. In: *Desktop planning: microcomputer applications for infrastructure and services planning and management.* eds. P. W. Newton, M. A. P. Taylor and R. Sharpe, Hargreen, Melbourne.

Jansenn, R. (1991) *Multiobjective decision support for environmental problems.* Vrije Universiteit te Amsterdam, Druckkerij Elinkwijk.

Kangas, J. and Kuusipalo, J. (1993) Integrating biodiversity into forest management planning and decision-making. Forest Ecology and Management. 61:1-15.

Kuusipalo, J. and Kangas, J. (1994) Managing biodiversity in a forestry environment. *Conserv. Biol.* 8: 450-460.

Kuik, O. and Verbruggen (1991) In search f indicators of sustainable development. Kluwer Academic, Dordrecht.

Lubchenco, J. et al (1991) The sustainable biosphere initiative: an ecological research agenda: a report from the Ecological society of America. Ecology 72:371-412.

Lutz, E. and Munasinghe, M. (1994) Integration of environmental concerns into economic analyses of projects and policies in an operational context. Ecol. Econ. 10:37-46.

McNeely, J. A. (1988) Economics and Biological Diversity: Developing and using economic incentives to conserve biological resources. IUCN, Gland, Switzerland.

Munasinghe, M. (1993) Environmental economics and sustainable development. World Bank Environment Paper Number 3. The World Bank, Washington, D.C.

Munda, G., Nijkamp, P. and Rietveld, P. (1994) Qualitative multi-criteria evaluation for environmental management. Ecol. Econ. 10: 97-112.

Munda, G. (1993) Multiple-criteria decision aid: some epistemological considerations. J. Multi-criteria Decision Analysis 2:41-55.

Nijkamp, P. (1979) Multidimensional spatial data and decision analysis. Wiley, New York.

Pearce, D., Markandya, A. and Barbier, E. B. (1989) Blueprint for a green economy. Earthscan Publications, London.

Pearce, D. and D. Moran (1994) The economic value of biodiversity. Earthscan, London.

Pearce, D. and Turner, R. K. (1990) Economics of natural resources and the environment. Harvestor Wheatsheaf, New York.

Petry, F. (1990) Who is afraid of choices? A proposal for multi-criteria analysis as a tool for decision-making support in development planning. J. Intern. Devel. 2:209-231.

Pressey, R. L. Humphries, C. J., Margules, C. R., Vane-Wright, R. I. and Williams, P. H. (1993) Beyond opportunism: key principles for systematic reserve selection. *Trends Ecol. Evol.* **8**, 124-128.

Romero, C. and Rehman, T. (1987) Natural resource management and the use of multiple criteria decision-making techniques: a review. Euro. Rev. Agr. Econ. 14:61-89.

Sokal, R. R. and Sneath, P. (1973) Numerical taxonomy. Freeman. San Francisco.

Spellerberg, I. F. (1992) Evaluation and assessment for conservation. Chapman and Hall, London

Tietenberg, T. H. (1990) Economic incentive policies and sustainability. *in* (B. Aniansson and U. Svedin, eds.) Towards an ecologically sustainable economy. Swedish Council for Planning and Coordination of Research.

Turner, R. K. (1990) Sustainability, resource conservation and pollution control: an overview. in: (R. K. Turner, ed.) Sustainable environmental management: principles and practice. Belhaven Press.pp. 1-25.