Environmental Conservation 28 (2): 104–116 © 2001 Foundation for Environmental Conservation

The safe minimum standard of conservation and endangered species: a review

ROBERT P. BERRENS*
Department of Economics, University of New Mexico, Albuquerque, NM 87131, USA
Date submitted : 5 July 2000 Date accepted : 26 January 2001

Summary

The safe minimum standard (SMS) approach is a collective choice process that prescribes protecting a minimum level of a renewable natural resource unless the social costs of doing so are somehow excessive or intolerably high. Arguments for the SMS are typically invoked in settings involving considerable uncertainty and potentially irreversible losses. However, the SMS is most commonly viewed as existing only on the periphery of thought in traditional environmental and resource economics. The specific objectives are: (1) to define the SMS approach generally and examine theoretical support, particularly for its application in endangered species decision settings; (2) to examine the relationship between an SMS approach and benefit-cost analysis (BCA); (3) to examine the relationship between an SMS approach and non-market valuation; (4) to compare an SMS approach to alternative definitions of sustainability; and (5) to review the general consistency of the SMS approach with the USA's Endangered Species Act (ESA) of 1973, as amended. Recent attention on this pragmatic policy approach has been something far greater than cursory, with advances and detailed discussions on theoretical considerations, philosophical underpinnings and case study applications. While the SMS emerges as a fairly coarse policy instrument, its pragmatic value is seen in complex environmental policy applications, such as endangered species protection.

Keywords: safe minimum standards, endangered species, sustainability

Introduction

Finding social solutions to complex environmental problems involving long time frames, potentially irreversible losses and a high degree of uncertainty, requires pluralistic and pragmatic collective choice processes. Pragmatism refers to the philosophy that 'the value of a belief must be evaluated by the consequences of actions taken as a result of that belief', and pluralism refers to the 'use of multiple viewpoints or intellectual approaches when a complex social problem is subjected to analysis' (Castle 1996). In contrast, fully synoptic, once-and-for-all decision models may have more textbook appeal. But, in settings such as the protection of biodiversity and preservation of endangered and at-risk species, such approaches are unlikely to have the necessary information, to be flexible to changing circumstances over long time frames, or be practical in pluralistic public-policy settings. An alternative, pragmatic collective choice rule is the safe minimum standard (SMS) approach to the protection of a renewable resource under situations of potentially irreversible losses, in other words extinction.

The SMS approach is generally considered as something of an academic footnote, existing on the periphery of environmental and resource economics. Originally advanced by Ciriacy-Wantrup (1952) as the 'safe minimum standard of conservation', the SMS approach has been undergoing a sustained revival (Vaughn 1997). Within competing schools of thought in environmental and resource economics, Ciriacy-Wantrup's work and methodological approach clearly fall within the institutionalist tradition, focusing on how the rules of the game affect economic and social outcomes, with a strong policy-oriented pragmatism (Randall 1985). Ciriacy-Wantrup wrote widely on a variety of topics, including western water law in the USA, common property resources, and applications of benefit-cost analysis (BCA). Of particular note, in addition to introducing the SMS approach, Ciriacy-Wantrup (1947) was also the originator of the idea to use surveys to elicit the value of non-market goods from individuals, a technique now known as contingent valuation (CV; Cummings *et al.* 1986; Bateman & Willis 1999). The validity of CV for estimating preservation benefits has been a controversial point in discussions on the economics of endangered species.

Combined with this renewed attention on the SMS approach, recent developments in this literature merit review and detailed articulation to an environmental policy audience. The aim of this paper is to review the case for an SMS approach to protecting endangered species. The context and orientation for the review are primarily focused on the USA, but it involves issues that are found in numerous settings. Such a review and articulation is justified in that the SMS was always intended to be a pragmatic public policy tool, rather than kept on the shelf until all questions were resolved. The specific objectives of this review are: (1) to define the SMS approach generally and examine theoretical support, particularly for its application in endangered species decision

* Correspondence: Dr Robert P. Berrens Tel: +1 505 2779004 Fax: +1 505 2779445 e-mail: rberrens@unm.edu

https://doi.org/10.1017/S037689290100011X Published online by Cambridge University Press

settings; (2) to examine the relationship between an SMS approach and benefit-cost analysis (BCA); (3) to examine the relationship between an SMS approach and non-market valuation; (4) to compare an SMS approach to alternative definitions of sustainability; and (5) to review the general consistency of the SMS approach with the USA's Endangered Species Act (ESA) of 1973, as amended. The argument submitted in this review is that recent attention on this pragmatic policy approach has been something far greater than cursory, with advances and detailed discussions on theoretical considerations, philosophical underpinnings and case study applications. While the SMS emerges as a fairly coarse policy instrument, its pragmatic value is seen in complex environmental policy applications, such as endangered species protection.

The SMS approach: definition and background

Defining the SMS

Simply stated, the SMS approach requires that some safe minimum level of a renewable natural resource be protected unless the social costs of doing so are somehow 'excessive', 'intolerable', or 'unacceptably large'. The notion of what constitutes intolerable is to be decided by the political or administrative process in any particular case (Batie 1988; Castle 1996). Leaving this determination to a case-by-case basis has led many critics to dismiss the SMS approach as uselessly vague and 'fuzzy'. Investigation of the types of considerations that might constitute intolerable cost is critical to the SMS concept being accepted as a pragmatic policy tool. Following Randall (1991), invoking the intolerable cost argument to circumvent a preservation action ought to require some 'extraordinary decision process'. For example, more than one standard might be required, or more than one level of decision-making and separation of function might be involved.

The notion of intolerable costs might be extended to include not only the aggregate level of costs, but also an inequitable distribution of those costs. For example, many endangered species preservation questions involve a classic spatial mismatch of highly concentrated local costs against widely dispersed benefits of preservation (Brown & Shogren 1998). Endangered species debates are often as much about the distribution of economic consequences as they are about the size of those consequences (Polasky *et al.* 1997; Shogren & Hayward 1997). Randall and Farmer (1995) acknowledged that an SMS approach might be amended to included distributional considerations. Thus, the consensus process for defining acceptable costs is likely to include distributional considerations (Berrens *et al.* 1999). Distributional effects constitute a valid political and moral component of the economic impacts of species preservation.

Recognizing distributional concerns opens consideration to questions of compensation and mitigation. The dimensions of such concerns include geographic regions, specific economic sectors, particular social groups (e.g. native American tribes or other indigenous peoples) and individual land owners. The question of appropriate compensation to particular individuals who must bear the localized costs of species preservation can be viewed as part of the collective choice process for determining tolerable social costs, and their distribution. Compensation and mitigation can occur within negotiations over a particular preservation case, as sometimes happens in ESA cases in the USA (e.g. Berrens *et al.* 1999), or may be specified by some larger policy framework, such as the European Union's (EU) Habitats Directive and the inter-related Common Agricultural Policy (CAP). For example, in the CAP, a core principle is that farmers should be expected to meet basic environmental thresholds, but for provision of environmental services above this baseline, such as special habitat protection, direct compensation for any lost income or costs is expected (European Commission 1999).

Theoretical support for the SMS

The SMS was first proposed by Ciriacy-Wantrup (1952) as a flexible policy tool to help protect renewable natural resources; he was not concerned with substantially extending the theory of optimal social choice. Rather, Ciriacy-Wantrup (1952) was concerned with developing a pragmatic tool for collective choices in the face of high degrees of uncertainty and limited scientific information, and potentially irreversible losses (Castle *et al.* 1996). However, there is considerable theoretical support for justifying such a pragmatic policy tool.

Well-known attempts have been made to operationalize the SMS in game-theoretic terms (Bishop 1978; Tisdell 1990; Ready & Bishop 1991). These attempts have focused not on risk, but simple games of preservation versus development choices involving a type of pure uncertainty, that is well-defined outcomes with known payoffs but unknown probabilities (Palmini 1999). The evolution of this literature is insightful.

Bishop (1978) originally posited a simple model of a game against nature, referred to as the 'insurance game', where society is uncertain about whether or not a disease will occur. The cure for the disease is known to be found in a natural species, but there is true uncertainty about whether the disease will occur. Society's mutually exclusive choices are either preservation of the species, or development (with the irreversible loss of the species and the cure). If society were to follow a minimax decision rule, and minimize the maximum possible losses, then the preservation strategy would be chosen. Bishop (1978) proposed a 'modified minimax rule' that recognized the social opportunity costs of forgone development, and argued for following a SMS rule of choosing preservation unless the social costs of doing so were 'unacceptably large'.

An alternative to the insurance game, is the 'lottery game', first presented by Ready and Bishop (1991), where there is certainty that a disease will occur but uncertainty in whether the cure will be found in a natural species. In this alternative formulation of the game against nature, following the same minimax decision rule (minimize maximum possible losses) does not lead to the choice of the preservation strategy. Thus, in comparing the insurance and lottery games, Ready and Bishop (1991) argued that the predictions of game theoretic models are ambiguous because results are highly sensitive to the initial framing of the game against nature. Left with this ambiguity, they concluded that the SMS may be without rigorous theoretical foundations and yet may still yield the 'right' societal choice. The implication is that support for the SMS approach can only be based on appeals to particular moral arguments or value judgements.

In a reassessment of previous SMS game theory investigations, Palmini (1999) argued that a minimax regret criterion, selecting a strategy that minimizes the maximum possible regret of the wrong choice, is appropriate and provides unambiguous support for an SMS strategy in species preservation choices (i.e., either an insurance game or lottery game). Regret can be thought of as the welfare loss from choosing an action and then seeing the other state of the world or game occur. Under the mimimax regret criterion, society chooses the alternative that costs it the least reduction in welfare if the wrong choice is made. In the lottery game, if society chooses development and a cure was indeed available, it would have foregone a cure for a possibly disastrous disease. Alternatively, the cost of choosing preservation (and then not observing a cure) would be foregone development benefits minus any preservation benefits (e.g. non-market amenities, etc.). The preferred choice depends on any assumptions embedded in the game about the relative sizes of the payoffs of different outcomes. But Ciraicy-Wantrup (1952) clearly expected that foresaken development benefits would not typically exceed the benefits of preservation. Support for this assumption is found in the relatively few BCA studies that have been conducted for endangered species, such as Rubin *et al.* (1991) and Hagen *et al.* (1992) for the preservation of the northern spotted owl in the USA's Pacific Northwest. Even if this were not the case (say, preservation barely failed a benefit-cost test), modest potential for irreversible loss would still justify preservation. The point is that in situations of true uncertainty, risk-averse agents may rationally adopt minimax regret decision rules.

Thus, the SMS approach can be simply cast as a strategy for avoiding regret in situations where physical parameters are poorly understood and there is a potentially irreversible loss. In such situations a rational decision criteria may be to consider the costs of being wrong, and under such a minimax regret decision rule, the SMS is consistent in both the lottery and insurance games. In addition to this recent game-theoretic support, other related theoretical developments in support of the SMS are also emerging.

Drawing from earlier work in economics on axiomatic rational choice models, Woodward and Bishop (1997) argued that in cases of true (Knightian) uncertainty, decision makers may attach great weight to avoiding worst case scenarios. They characterized this as the 'expert panel problem', where there is fundamental disagreement across a panel of experts about some physical outcome (e.g. probability of extinction across different habitats). Specifically, in decision settings that involve catastrophic and irreversible outcomes, and where there is no meaningful way to assign a probability distribution to possible outcomes, then it may be quite rational to focus on endpoints of the possible outcome space rather than some midpoint.

The implication of this work is that it may be quite rational for public-policy decision makers to adopt precautionary strategies, such as the SMS (Arrow *et al.* 2000). Woodward and Bishop (1997) explicitly identified the SMS approach of Ciriacy-Wantrup as one such strategy, and the protection of endangered species under the ESA as a relevant policy setting. By extension then, the adoption of an SMS approach within legislation such as the ESA, may be argued to have a rational economic basis.

For efficiency reasons environmental economists generally prefer incentive-based approaches over quantity-based approaches to environmental regulation; for example, pollution taxes will be preferred to pollution quantity standards. However, there are exceptions to these arguments. In a well-known theoretical piece on the economics of environmental regulation, Weitzman (1974) demonstrated that physical quantity controls (standards) may be preferred to prices (i.e., pollution taxes) if there is relatively substantial uncertainty around the benefits function (i.e., the benefits of environmental protection). Specifically, this might be the case if there were relatively greater uncertainty about the benefits of an environmental protection action relative to the costs (i.e., forgone development benefits) of a protection action.

Such arguments would seem to connect to controversial attempts to value the monetary benefits of endangered species protection, which might be largely composed of non-use values (purely contemplative values in the absence of any direct *in situ* use value). In the absence of reliable benefit information, a standard-based approach to environmental regulation may be theoretically justified. Rather than problems with monetization of preservation benefits, some sources argue that the core difficulty with identifying a benefits function lies with 'tracing out' the physical consequences of some economic activity (e.g. Hanemann 1995). Specifically, it is the pervasive complexity and stochasticity of natural ecosystems that makes tracing out the physical (non-monetary) damage function so difficult (Arrow *et al.* 2000). Further, there is usually very limited opportunity for controlled experiment at the necessary landscape/ecosystem level of scale; this may be especially true in the case of habitats for at-risk species.

All these arguments then raise the question of how we would go about trading off benefits and costs to find the 'optimal' level of species preservation, when there is substantial uncertainty around a benefits function. In the specific context of environmental pollution, Hanemann (1995) answers by pointing toward an SMS approach: 'The answer depends crucially on how one conceptualizes the uncertainty, and on one's attitude toward risk. If one sees the risks in terms of potential thresholds and nonlinearities in the benefit function around a "safe minimum standard", not only does this strengthen arguments for quantity controls . . . but it also may argue for allowing a lower level of emissions for safety's sake. Regulator's reluctance to trade off benefits and costs as fully as economists would like may represent a legitimate but risk-averse response'. Thus, similar to the arguments reviewed from game theory and expert choice problems, in selected decision settings there may be theoretical support for an SMS strategy.

Applying the SMS to endangered species protection

The initial element of the SMS approach involves identifying a critical biological threshold. An important point of clarification is that the SMS does not refer just to a simple physical safety standard, but rather to a larger collective choice process. But, certainly, part of this larger process is defining the safety standard. Common candidates for physical thresholds include minimum instream flows (Berrens *et al.* 1998), and maximum tolerance for soil loss (Schaeffer & Cox 1992). For endangered species, possible thresholds would include minimum viable populations (Soulé 1987) or habitat areas (Bishop & Woodward 2000)

In the context of endangered species protection, some authors have argued that the SMS cannot be operationalized; for example, consensus as to what constitutes a minimum viable population and therefore the minimum habitat needed by a species is often lacking (Hohl & Tisdell 1993). The implication is that there is too much uncertainty around the safety standard itself, however defined. While biologists and ecologists may often be able to provide only approximations and rules of thumb (Hohl & Tisdell 1993), the SMS is purposefully designed to be a pragmatic approach. While not diminishing the difficulty of establishing physical thresholds, this only underscores the point that an SMS approach is likely to be more robust than decision-making processes resting on the notion of fully synoptic assessments, including BCA.

While the SMS has traditionally been discussed in terms of single species, the concept is not tied to this. For example, the SMS approach could be applied to a measure of physical habitat or number of hectares preserved (Rodgers & Sinden 1994), or to emerging indices of biodiversity (Solow & Polasky 1994; Metrick & Weitzman 1998). SMS approaches have been applied in various case studies to constellations of associated species, such as multiple fish species with similar riverine habitat requirements (Berrens *et al.* 1998).

Of course, consideration of endangered species is also part of the larger picture of protecting biodiversity and functioning ecosystems. In its recent position piece on strengthening the USA's ESA, the Ecological Society of America endorsed the use of procedures such as population viability analysis in determining minimum viable populations and critical habitats for individual species, while recognizing the difficulties involved and encouraging an ecosystem orientation (Ecological Society of America 1995). Recent implementation of the ESA by the US Fish and Wildlife Service has also tried to encourage multi-species habitat protection and ecosystem preservation (Solow & Polasky 1999). Norton and Ulanowicz (1992) argued for a hierarchical approach to protecting biodiversity and natural systems, where different levels have different rates of change. Biodiversity policy must be implemented at the landscape level, where individual species contribute to the larger community. A central aim is to protect as many species as possible, but perhaps not all if the costs are simply too high. None of these arguments is inconsistent with an SMS approach to protecting endangered species, which can be a component of a larger biodiversity policy.

Relationship of the SMS approach to benefit–cost analysis

Traditional economic approaches, such as benefit-cost analysis (BCA), to collective choice questions have their philosophical roots in utilitarianism. Further, the normative basis of utilitarianism is considered to have several basic elements, including consequentialism and welfarism (Sen 1987; Hamlin 1989). Consequentialism evaluates all policy choices solely on the basis of their consequences (outcomes) for alternative social states. Welfarism implies that the evaluation of social states is done solely on the basis of individual utility or value information (that is other information is treated at best as the raw material for individual values).

Criticisms of consequentialism and welfarism take on particular strength in situations involving true uncertainty and potential irreversibility, and where welfarist information may be missing or incomplete, as in the case of market values of a particular environmental service. Welfarism may also conflict with right-based approaches in particular decision settings (Sen 1987; Berrens & Polasky 1995).

While generally left unstated, the basic utilitarian assumptions are the foundation for much of the applied work in environmental economics. Maximizing the present-value of aggregated net-benefits, or 'welfare maximization' remains a prevalent, if not the dominant, economic perspective for approaching questions of biodiversity and species preservation.

In evaluating a particular preservation project, BCA provides the decision criteria for determining whether or not to proceed. Some authors distinguish BCA when it is simply an information system, rather than an explicit decision rule or collective choice process (Lesser & Zerbe 1995). But, the practical context of endangered species policy requires that a collective choice process be articulated. Further, some proposed 'economic' reforms to the ESA have focused on the requirement that preservation actions pass a net-benefits test;

for example, at least one recent congressional proposal (House of Representatives Bill 1490, 103 Congress, 1st Session, 1993) would amend the ESA to require BCA of critical habitat designation (Berrens *et al.* 1998).

The basic premise of BCA is that if aggregated individual benefits of a preservation action outweigh the aggregated individual costs, in present value terms, social welfare is increased. In the case of negative net benefits, society is seen to gain from forgoing the species preservation action. As Brown and Szweirbinski (1988, p. 91) argued: 'Not all species should be preserved; we should actively seek to preserve only those for which the expected net benefits are positive'. Setting aside normative considerations for the moment, the implication of such a statement is that the economic tools of BCA are up to the job of measuring the net benefits for all species preservation actions. Many economists would argue that we can do just that, if just given the opportunity and the resources. However, the evidence supporting this argument is much more mixed (Bateman & Willis 1999), as will be reviewed in the following section.

The use of BCA has been criticized extensively in the context of preserving biodiversity and at-risk species (Norton 1987). Criticisms include the problem of determining the social discount rate, capturing ecosystem complexity, accurate valuation of non-market benefits including existence values and identification of all consequences, as well as philosophical criticisms of the utilitarian framework (Sagoff 1988; Hanley 1992; Hubin 1994).

Since they are irreversible, species losses involve intergenerational equity issues by constricting the choice sets of future generations (Perrings 1994). BCA decision rules neglect such fundamental issues as the intergenerational allocation of natural endowments. Explicitly raising the intergenerational equity issue for any BCA involving an endangered species, Bishop (1980) restated the decision problem as: 'To what extent is it fair for the current generation to bear costs in order to reduce uncertainty faced by future generations?' The converse question can also be asked: to what extent is it fair for the current generation to avoid costs in order to increase uncertainty faced by future generations? An efficiency-oriented approach would completely overlook this ethical issue. Thus, the current generation can disadvantage future generations through actions that affect the endowment bundles, and it may be necessary to go beyond efficiency criteria in the case of preserving endangered species (Bishop 1993).

Some authors have argued that BCA at least must be augmented by additional sustainability constraints (Pearce 1976; Hanley 1992; Toman 1994) for decisions involving long time horizons, true (Knightian) uncertainty, and potentially irreversible changes. The scale and motivation for such offered constraints will differ greatly. In the case of protecting biodiversity, micro-level standards can be motivated by concern for discontinuities and threshold effects in complex ecological systems (Perrings & Pearce 1994).

In response to perceived limitations of BCA, especially in the context of preserving at-risk species and biodiversity, the SMS is frequently suggested as an alternative collective choice rule. The SMS can perhaps best be conceptualized as a burden-of-proof switching device (Batie 1988; Tisdell 1990). While conventional economic analysis strives to determine the net benefits of preservation actions, SMS starts with the assumption that preservation of an endangered species is *a priori* beneficial, but remains sensitive to the social costs of any preservation action. Scott (1999) likened the SMS to legal trust doctrine, which emphasizes the preservation of assets and calls for special caution in conditions of uncertainty. The burden-of-proof lies in demonstrating that the opportunity costs of preservation actions are intolerable. Determination of intolerable costs is crucial in implementing the SMS, and entails a larger collective choice process beyond demarcating a simple physical standard (Norton 1987).

In contrast to the standard BCA decision rule, SMS is not a fully welfarist approach in that it does not require complete estimation or articulation of the monetary benefits of preservation. Crowards (1998, 1999) has argued that the concept of the SMS can be a means of supplementing purely reductionist decision tools to incorporate ethical concerns into decision making. Similarly, Randall and Farmer (1995) argued that viewed from multiple philosophical lenses (utilitarian, rights- or duty-based, contractarian) there is a strong but circumstantial case for conserving biodiversity. A failsafe defence of actions to protect biodiversity and at-risk species in all circumstances at any cost may be difficult to defend; there is no trump card over other moral concerns. However, an approach that considered the benefits and costs of preservation actions subject to an SMS constraint would be amenable to pluralistic philosophical perspectives (Randall & Farmer 1995; Castle 1996; Farmer & Randall 1998).

Relationship of the SMS approach to non-market valuation

Non-market valuation refers to the assessment of economic values for goods and services that are not priced and traded in a market, for example outdoor recreation, or species and wilderness preservation. Absence of a functioning market does not imply the absence of economic value, but it does complicate its assessment in BCA and natural resource damage assessments. Considerable effort has been invested over the last three decades into developing a battery of techniques for measuring non-market values of individuals for environmental goods. As Arrow (1994, p. 1) noted: 'The typical economist's argument today for government intervention to protect the environment rests on individual valuation'.

In any typology of values for non-market goods the critical distinction is between use values, such as for outdoor recreation, and non-use values. Non-use values are purely contemplative values that by definition have no discernible trail to market behaviour (Carson *et al.* 1999). Perhaps the archetypical non-use value is existence value, which is argued to arise from 'simply knowing that some desirable thing or state of affairs exists' (Randall 1991).

Non-market values for endangered species may be heavily motivated by non-use values. The assessment of such values is dependent upon the survey-based contingent valuation (CV) method. Professional opinion of the validity of CV for measuring non-use values remains mixed (Diamond & Hausman 1994; Hanemann 1994). For example, in the case of applying CV to single species, respondents may have trouble isolating values from the larger habitat or general preservation program (Stevens *et al.* 1991). A conceptual alternative is to value broader ecological composites such as biodiversity protection, but this may suffer from the lack of a precisely defined commodity (Vatn & Bromley 1995).

In a recent statistical meta-analysis of endangered species valuation results from over 25 different CV studies, Loomis and White (1996) presented evidence that there is important systematic information about the social benefits of protecting endangered species. However, recognizing potential validity problems they stopped short of endorsing the use of such values in strict benefit-cost decision rules, and instead supported SMS approaches in collective choice rules to protect endangered species (Loomis & White 1996).

The SMS approach is not dependent on the valuation of the non-market benefits of species preservation. The SMS approach has been viewed ambiguously as both a substitute for the estimation of existence values for endangered species, or as a complement to such measurements (Berrens 1996). The latter perspective is adopted here; attempts to measure existence values and the adoption of an SMS strategy are complementary. Specifically, when reinforced by an SMS decision rule, continued refinement of the measurement of the non-market benefits of species preservation may help provide relative information in a kind of 'gross disproportionality' test with other social benefits and costs (Randall & Farmer 1995). While non-market benefits could be included within an SMS approach (Bishop & Woodward 2000), this is something short of saying that defensible estimation of existence values is obligatory within an SMS approach. Crowards (1998, 1999) presented a contrasting view, arguing that non-market benefits estimation, including that of non-use values, should be obligatory within an SMS approach. While we can be sympathetic to the argument, the point Crowards (1998, 1999) missed was that the devil is in the details. It is not just that CV estimates of non-use values may be imprecise. CV responses in an endangered species context may violate validity tests, at least for some portion of the population who may not accept the implied trade off (Turner 1999).

It is unclear whether Ciriacy-Wantrup would have approved of the controversial extension of CV techniques into the measurement of non-use values (a practice that post-dates his career), and further requiring their consideration in a SMS strategy. To wit, Ciriacy-Wantrup (1961) wrote of his concern with extending quantitative techniques beyond their limit. However, Ciriacy-Wantrup (1955) also saw an inherent process value in persistent attempts at quantification; they provide a potential check on the arguments of vocal interests and may have a stimulating effect in expanding scientific understanding of all dimensions of environmental policy. The further constraint that he placed on monetary quantification was that he reserved a primary role for physical indices (e.g. safe minimum standards; Berrens 1996).

From a pragmatic perspective, much can be learned from the attempt to require valid estimation of non-use values into rule-making for natural resource damage assessment (NRDA) and liability cases in the USA. In the policy debate of the 1990s, a prominent panel convened by the National Oceanic and Atmospheric Administration (NOAA) gave a conditional endorsement to the use of CV to measure non-use values, along with a set of suggested guidelines (Arrow *et al.* 1993). However, many other prominent economists went on record in severely criticizing attempts to measure fully such values. This is perhaps best evinced by debates over the inclusion of non-use values in the NRDA for the Exxon Valdez case (Hausman 1993; Diamond & Hausman 1994). In the face of continuing criticisms, the NOAA panel endorsement and guidelines were not taken up in subsequent regulatory rule-making (Jones & Pease 1997). In the end, application of CV is allowed but not required as an input in an NRDA case. This is not dissimilar to what is seen in some endangered species cases (Walsh 1992; US Army Corps of Engineers 2000).

Within the context of the NRDA debate, considerable criticism was made of the CV method for measuring non-use values (Hausman 1993), yet little in the way of alternative decision rules was offered. Shavell (1993) presented an exception, counselling that there should be no worry that non-use values for the environment will be ignored; the legislative process can provide direct protection when needed. The specific example given was the USA's Endangered Species Act. This recommendation can be juxtaposed against that of other economists who have criticized the current structure of the ESA for not paying enough attention to the weighting of benefits and costs (Shogren & Hayward 1997; Brown & Shogren 1998).

Despite many theoretical and empirical advances, the debate among economists over our ability to measure validly non-market values, and especially the non-use values that may be prevalent in endangered species contexts, now stretches over several decades and shows little sign of abating. As Solow and Polasky (1999, p. 21) stated, 'Despite the great effort to develop and apply the tools of non-market valuation, it is not clear that it will ever be possible to get a reliable objective estimate of the worth of a species'. The absence of such information, or of a general consensus among economists on how to get it, makes appeals to utilitarian-based approaches (e.g. BCA) to species preservation sound hollow. While strict welfarist approaches can be criticized from a variety of philosophical perspectives, they may be particularly unappealing when the individual value information is seriously incomplete (Sen 1987).

Relationship of the SMS approach to sustainability concepts

It is valuable to step back from debates over measurement of non-market values and ask about the implications for sustainability even if non-market values were measured perfectly. As now recognized, even perfect measurement provides no guarantee that an environmental sustainability constraint (however defined) will be met (Howarth & Norgaard 1992; Bishop & Woodward 2000).

Few topics have received greater public and scholarly debate over the last decade than that of sustainability and sustainable development. Perhaps the most commonly referenced definition is that proposed by the United Nations' World Commission on Environmental Development (WCED 1987): 'Sustainable development is development that meets the needs of the present without jeopardizing those of the future'. Recognizing sustainability as involving obligations to the future raises the question of what shape this obligation takes, and there is a full spectrum of perspectives on sustainability. The specific concern here is the relationship of an SMS approach to different perspectives.

For some sustainability writers (e.g. Pezzey 1992), the obligation to the future takes the form of non-declining social utility (U):

$$U_{t+1} \geqslant U_t \tag{1}$$

That is, the utility of the future generation (U_{t+1}) should be at least as great as that of the current generation (U_t). The link to the further future is made in the chain of obligation from one generation to the next. The immediate practical problem with trying to implement a sustainablity constraint like Equation (1) is the impossibility of measuring an aggregate generational utility measure. Attention is quickly directed to common economic proxies, such as non-declining consumption (C) expenditures or income measures:

$$C_{t+1} \geqslant C_t \tag{2}$$

where C_t represents the consumption expenditures of the current generation, and C_{t+1} represents the consumption expenditures of the subsequent generation. An alternative perspective is to emphasize the capital investment perspective in sustainability debates. Following Solow (1992), a conceptual link can be made from protecting consumption opportunities through protecting society's aggregate capital stock, and thus protecting future choices. While recognizing underlying moral concerns (Solow 1992), discussions of sustainability must be concerned with investments in maintaining society's capital stock.

Alternative sustainability perspectives can be differentiated by the amount of structure they would impose on the capital bequest package that is turned over in the chain of obligation to the future (Norton 1995; Turner 1999). More formally, consider the simplified expression:

$$TK = MK + TNK + SK \tag{3}$$

where TK represents society's total capital stock, composed broadly of man-made capital (MK), total natural capital (TNK), and cultural or social capital (SK). Each composite capital stock might be thought of as a vector of individual elements, which might be further broken down. Specifically of interest is the decomposition: $TNK = RNK + NNK$, where the distinction is made between renewable natural capital (RNK) and non-renewable natural capital (NNK). However, the broad decomposition in Equation (3) allows us to differentiate alternative sustainability perspectives in the terms of the constraints that might be imposed on economic development.

A now-common distinction on the spectrum of sustainability perspectives is between weak sustainability, which emphasizes the provision of a non-declining total capital stock for society, and strong sustainability, which emphasizes a non-declining total natural capital constraint (see Folke *et al.* 1994). Variants of the strong sustainability position also introduce the notion of identifying and protecting critical natural capital assets (Turner 1999).

Weak sustainability advocates might impose the condition that,

$$TK_{t+1} \geqslant TK_t \tag{4}$$

where TK_t represents the total capital stock of the current generation, and TK_{t+1} represents the total capital stock of the subsequent generation. This implies unlimited substitution among the composite categories. Implementing such substitutability implies comparable units of measurement, typically chosen to be a monetary index (i.e. \$TK). The most daunting informational task may be valuing non-market environmental assets (subsumed in TNK). It is worth emphasizing that economic values for the environment are essentially indices of substitution; they show how we can trade environmental services for money. As reviewed in the previous section, recent debates among economists concerning the possibility of validly and reliably estimating non-use values for environmental preservation only highlight the difficulty in making certain substitutability assumptions.

At the other end of the continuum, strong sustainability might impose the condition that:

$$TNK_{t+1} \geqslant TNK_t \tag{5}$$

where TNK_t represents the stock of total natural capital in the current generation, and TNK_{t+1} represents the stock of total natural capital in the subsequent generation. While more restrictive than Equation (4), Equation (5) still implies substitution possibilities among types of natural capital. Thus, a further restrictive condition might be expressed as a set (A) of non-declining constraints on development protecting elements (a) of critical natural capital:

https://doi.org/10.1017/S037689290100011X Published online by Cambridge University Press

$$aNK_{t+1} \geqslant aNK_t, \forall a \text{ where } a = (1, 2, ., ., ., A) \quad (6)$$

where aNK_t represents the stock of a critical natural capital asset in the current generation, and aNK_{t+1} represents the stock of a critical natural capital asset in the subsequent generation. Constraints (5) and (6) might theoretically be expressed in monetary units, but more typically in strong sustainability arguments they are depicted as physical constraints. For example, the set (A) in Equation (6) might include as a proper subset a list of minimum instream flow requirements, at specific times and locations within a river system.

Differences between alternative perspectives on sustainability hinge on key underlying assumptions, the two most prominent being the substitution relationship between man-made and natural capital, and renewable natural capital and non-renewable natural capital (Costanza & Daly 1992; Daly 1994; Norton 1995). Those who believe that substitution possibilities are limited will support a strong sustainability criterion and the protection of natural capital, especially capital assets which may possess high degrees of genetic diversity, or loss of which may be potentially irreversible, as in the case of endangered species.

While the concept of the SMS in resource and environmental economics predates the emergence of debates on sustainable development and sustainability, it can be viewed within the context of 'sustainability constraints' (Bishop & Woodward 2000). For example, individual elements within Equation (6) would appear to have an obvious connection to safe minimum standards. In implicitly accepting some notion of critical natural capital, the SMS approach has been recognized by some authors as being operationally equivalent to a structured social bequest, and thus connected to weak sustainability perspective. However, the SMS approach has been viewed as falling somewhere between weak and strong sustainability criteria (Turner *et al.* 1994). Consistent with strong sustainability criteria, the SMS approach recognizes the imperative to protect critical natural capital (e.g. habitat for endangered species), but stops short in the conditional nature of the imperative, specifically, the sensitivity to the level of social costs. The SMS decision rule states that protection of the standard should be met unless the social costs of doing so are 'intolerably' high. That is, the safe minimum condition can be violated depending on the social cost of meeting that condition within the current economy. Thus, it retains the element of consequentialism.

At this point it is useful to return to Equation (3) and the consideration of protecting capital assets. Much has been made in the debates about sustainability over the degree of substitutability between man-made capital assets (MK) and natural capital (TNK), but the concept of social capital (SK) is also an important consideration for the bequest package that is left to the future (Folke *et al.* 1994). Social capital would include various cultural and educational dimensions, but would also include the institutional and legal entitlements left in place for public policy generally, and in the specific case of environmental policy. For endangered species issues, the social capital aspect of the sustainability question includes what rules of the game we want to leave in place for the next generation of decision makers. These rules will constitute the policy over that natural endowment. To this end, we turn to the consideration of a specific policy example, the USA's Endangered Species Act (ESA).

The SMS approach and the ESA

Originally passed in 1973, the ESA has been amended a number of times and is likely to continue to evolve. In its original form, the ESA was an extremely stringent piece of legislation and could be described as a Kantian 'categorical imperative' to protect listed species, without any consideration of benefits and costs. The US Supreme Court in its decision on Tennessee Valley Authority vs. Hill, stated that 'the plain language of the Act, buttressed by its legislative history, shows clearly that Congress viewed the value of the endangered species as incalculable' (see Rohlf 1989). This original categorical imperative to protect has been altered by a series of amendments to the ESA; in its place is what can be described as a more conditional or hypothetical imperative (Kant 1949). It is the ESA in its current form, as a conditional imperative, that can be linked to the SMS approach.

Reauthorization of the ESA has been pending since 1992, with subsequent implementation relying on yearly congressional appropriations (Ando 1998). Important elements in the debate include: (1) special consideration to the rights and incentives of private landowners (Polasky *et al.* 1997; Innes *et al.* 1998); (2) formal incorporation of a multi-species orientation to capture potential economies of scale in protection (Dobson *et al.* 1997); and (3) the role of the ESA within a larger biodiversity policy framework (Ecological Society of America 1995). Similar to the SMS approach itself, the question of whether and how economic benefit information concerning species preservation might be included in the ESA's economic analyses remains an important issue (Brown & Shogren 1998).

A variety of sources have noted that the general structure of the ESA, as amended, is consistent with the SMS approach (Foy 1990; Castle & Berrens 1993; Bishop & Woodward 2000). Thomas and Verner (1992) made a similar argument, albeit without explicitly identifying the SMS directly. While the initial orientation of the ESA is towards protection, information on economic consequences can be used at several different points in the process to modify or override this protection (Berrens *et al.* 1998; Innes *et al.* 1998; Solow & Polasky 1999).

A species under the protection of the ESA must be listed either as threatened or endangered. The listing determination is to be made utilizing the best scientific data available, and cannot be based on any economic analysis. Once a species has been listed, critical habitat for the species must be identified, and a recovery plan developed by the implementing agency (e.g. the US Fish and Wildlife Service; USFWS). Designation of final critical habitat is made after a draft economic analysis of the resulting impacts. Analysis of economic impacts begins with a biological evaluation to obtain the habitat requirements, which are converted into direct economic impacts by linking the resource requirements of the species to the economic activities that must be altered. A period of public comment then allows interested parties to provide additional evidence. The final economic analysis is used along with physical and biological data as inputs into the exclusion process. The ESA requires consideration of economic and other relevant impacts in determining whether to exclude proposed areas from critical habitat. Such exclusion cannot jeopardize the recovery of the listed species and be likely to cause extinction. A variety of consultations can happen throughout this process (Solow & Polasky 1999).

In implementation, the ESA directs that probable economic 'impacts' be considered in the exclusion process, and this necessitates economic modeling (e.g. input–output [I–O] or computable general equilibrium [CGE]). While falling short of true economic surplus measures, impact analyses are often the only practical measure of economic consequences, and will often be the centre of public debate in critical habitat cases (Berrens *et al.* 1998). Additionally, regional modelling decisions, for example the chosen region of analysis, can affect the accounting of aggregate impacts and their distribution.

In summary, the exclusion process under the ESA allows for the exemption of individual areas from designated critical habitat if inclusion would entail severe economic impacts. Comparably, the SMS would allow for the extinction of a species if the economic consequences of preservation were judged to be somehow intolerable, which might include distributional concerns. In practice, the ESA requires that criteria be developed for foregoing preservation actions, or excluding part of the critical habitat.

While the ESA permits exclusion if and only if doing so means the species is not threatened with extinction, it contains a second level exemption opportunity beyond the critical habitat exclusion process. This additional exemption opportunity comes under section 7 of the ESA in the form of specially requested hearings held by an exemption committee. Section 7 applies exclusively to federal agencies and requires that their actions not jeopardize the existence of the species, or destroy or adversely modify designated critical habitat (Rohlf 1989). However, under section 7, an appeal can be made to an Endangered Species Committee, which possesses the authority to permanently exempt species from ESA protection. The Committee is composed of federal cabinet-level members and appointed state representatives.

An Endangered Species Committee exemption can only be made after it is determined that development actions have no reasonable or prudent alternatives, and that it is in the 'public interest' to grant the exemption (Rohlf 1989). The Committee has the power to designate an Administrative Law Judge, secure federal agency information, hold hearings, subpoena witnesses, and allow cross-examination. There is explicit 'separation of function' in the exemption process from agency personnel involved in previous decisions involving the species (but who may still be called as witnesses). The type of information that might be included in a hearing could be wide ranging, and would be likely to include analysis of economic impacts and distributional concerns (Booth 1994). In practice, the critical habitat exclusion process is commonly used while the Committee exemption process has been rarely used (Booth 1994). Taken together, the critical habitat exclusion and Committee exemption processes are consistent with Randall's (1991) notion of an extraordinary decision-making process under the SMS.

While actual case studies of the SMS approach remain rare (Bishop 1980; Hyde 1989), there have been a number of recent developments. Berrens *et al.* (1998) investigated two case studies of endangered fish species in the Colorado River system, and argued that actions taken under the ESA were broadly consistent with an SMS approach. Results from USFWS implementation of a federal court order were used to detail this consistency. Specifically, in the exclusion process for designating final critical habitat, the lack of severe economic costs was determined by comparing expected aggregate regional economic impacts against historical fluctuations in the regional economy. Both case studies involved joint consideration of critical habitat for multiple species; thus, they were not limited to single species perspectives.

Despite the general consistency of these case studies with the SMS approach, a number of points should be made. First, considerably less than half of the more than 1000 listed species in the USA have designated critical habitat and recovery plans in place (Brown & Shogren 1998). Thus, the majority of listed species have not been through the SMS-type process outlined in Berrens *et al.* (1998). Second, the connection identified in Berrens *et al.* (1998) towards implementing an SMS approach fails to incorporate distributional concerns, which may be important in defining intolerable economic consequences of species preservation actions.

This line of argument has recently been evaluated in several prominent cases studies involving endangered species protection in the American west, which used results from various economic analyses conducted by or for the USFWS. Berrens *et al.* (1999) reviewed both the prominent northern spotted owl case in the Pacific Northwest (argued to be a keystone species) and endangered fishes in the San Juan River, Colorado. The regional models focus on the economic impacts (employment and income) from proposed preservation action. The two case studies differed considerably in modelling approaches. For example, they differed in the geographic scale of the analysis, and sectors of the economy analysed. Such choices are not innocuous in estimating economic impacts and their distribution. As a general statement, the more narrow the focus of the analysis, the greater the estimated impact; a broader focus tends to lessen the economic impact of preservation actions, since rather than being lost, resources have greater opportunity to be reallocated in the larger economy. One implication, observed in the case studies

(Berrens *et al.* 1999), is that there is likely to be a need for economic modelling conducted at several different levels. Additionally, the cases used different criteria for potential exclusion thresholds in defining critical habitat; neither provided detailed justification for the choice. No pre-defined guidelines currently exist, and this remains an important policy issue.

As detailed in Berrens *et al.* (1999), within the an SMS-type collective choice process considerable mitigation and compensation occurred in both the San Juan River and Pacific Northwest spotted owl case studies. However, both of these prominent USA case studies had considerable federal land and projects involved. Many endangered species cases and critical habitats will not be tied so closely to federal land and projects (Polasky *et al.* 1997), and for these situations considerations of mitigation and compensation become even more prominent. Failure to consider mitigation and compensation runs the risk of creating incentives for private landowners to eliminate species or habitat (Polasky *et al.* 1997).

Although often mentioned in terms of 'regulatory takings' under the Fifth Amendment of the US Constitution (e.g. Stroup 1997), neither the ESA nor current jurisprudence requires compensation for adverse impacts from ESA actions (Innes *et al.* 1998, p. 36). While a strict mandate for full compensation under a regulatory taking lacks theoretical support, it is argued that alterations to the ESA should give full consideration to carefully crafted compensation packages (Innes *et al.* 1998). For example, mitigation and compensation opportunities provide the incentives to private landowners to enter into habitat conservation plans (HCPs) under the ESA. As cooperative agreements with federal agencies that allow incidental take and habitat modification, and reduce the uncertainty of private landowners, development of HCPs has seen rapid growth since the mid-1990s (Solow & Polasky 1999). Mitigation and compensation actions need not be based on regulatory takings arguments (Bromley 1997). Rather they can be motivated by pragmatic considerations to entice participation by private landowners, and by pluralistic considerations for distributive justice and what might constitute intolerable costs within an SMS approach (Berrens *et al.* 1999).

Discussion and conclusions

It is difficult to think of collective choice rules where information on individual preferences would not be admissible (Randall 1991), but acknowledging that point is something different from requiring strict utilitarian/welfarist decision rules. For example, in situations characterized by extreme difficulty in measuring and aggregating preferences, and concerns for sustainability and the protection of critical natural capital for intergenerational endowments, the question turns to how to structure decision processes and what else should count as admissible information (Berrens & Polasky 1995). Within an SMS approach, this question is answered by saying that information on critical thresholds for environmental services is admissible not just in an instrumental way (e.g. an input to economic preferences), but in a fundamental way where constraints are violated only under great care (Berrens 1996).

One recent commentator summarizes the interest in Ciriacy-Wantrup's (1952) SMS approach as prompting some 'initial debate' in the late 1970s, and then existing largely on the 'periphery' of resource economics, with references largely limited to 'cursory' remarks (Crowards 1998). However, recent attention to this pragmatic policy approach has become far greater than cursory, with advances and detailed discussions on theoretical considerations, philosophical underpinnings and case study applications. To wit, reviews of the SMS approach are now common in environmental economics textbooks (Lesser e*t al.* 1997; Goodstein 1999; Chapman 2000). It has been recognized that the USA's Endangered Species Act, as amended, is generally consistent with the underlying principles of an SMS approach (Castle & Berrens 1993; Bishop & Woodward 2000). Thus, within the difficult political context of protecting endangered species, the SMS gives us footholds for thinking about the ESA.

Laws and regulations can be viewed as part of the institutional framework (along with social norms and informal rules) of collective actions that both constrain and liberate individual actions (Bromley 1989). However, they can also be viewed as revealed preference data about social choices. Thus, there is a dual perspective on the ESA. First, it can be viewed as the constraint on current action that potentially liberates future actions by protecting choice sets from irreversible change. Second, it also reveals an evolving compromise in the difficult social choice problem of protecting endangered species. That is, the legislation itself is a revealed social preference, an evolving consensus among pluralistic American values

From its original form as one of the most prohibitive pieces of environmental legislation ever enacted, the ESA has been significantly modified a number of times. Importantly, a number of those modifications have been targeted specifically to the inclusion of information about economic consequences and impacts. Within these considerations of economic information, there is nothing in the ESA that prevents the consideration of non-use values. However, to this point monetization of the benefits of species protection (e.g. through CV methods) has not been formally required. Similarly, attempts to base preservation decisions strictly on a benefit–cost test, have also been rejected. Given that economists have come to no clear consensus on whether non-use values can be validly measured, there is some rationality to such results.

Attempts to amend the ESA to seek strict preservation goals without exemptions or any consideration of economic consequences would appear to lack sufficient political will, and thus are neither pragmatic nor pluralistic. The same can be said of attempts to modify the ESA to force preservation actions to be judged solely on the basis of benefit-cost criteria. Farmer and Randall (1998) noted that there are simply no guarantees that society will always reach consensus concerning difficult preservation choices for protecting endangered species; however, where there is agreement, they argue that an SMS-type approach is likely to be involved. It is also likely that in coming to some consensus about tolerable costs of preservation, distributional issues such as mitigation and compensation will have to be addressed (Berrens *et al.* 1999). That a general level of philosophical consistency exists between the SMS approach and the current regulatory framework of the ESA, as amended, is neither irrational nor perhaps totally unexpected.

References

Ando, A. (1998) Ecosystems, interest groups, and endangered species act. *Resources* **130**: 7–9.

Arrow, K. (1994) Methodological individualism and social knowledge. *American Economic Review* **84**: 1–9.

Arrow, K., Daily, G., Dasgupta, P., Levin, S., Maler, K.-G., Maskin, E., Starrett, D., Sterner, T. & Tietenberg, T. (2000) Managing ecosystem services. *Environmental Science and Technology* **34**: 1401–1406.

Arrow, K., Solow, R., Leamer, E., Portney, P., Radner, R. & Schuman. H. (1993) Report of the NOAA panel on contingent valuation. *Federal Register* **58**: 4601–14.

Bateman, I. & Willis, K., eds. (1999) *Valuing Environmental Preferences*. Oxford, UK: Oxford University Press: 645 pp.

Batie, S. (1988) Sustainable development: Challenges to the profession of agricultural economics. *American Journal of Agricultural Economics* **75**: 1084–1101.

Berrens, R. (1996) The safe minimum standard approach: An alternative to measuring nonuse values for environmental assets? In: *Forestry, Economics and the Environment*, ed. W. Adamowicz, P. Boxall, M. Luckert, W. Phillips & W. White, pp. 195–211. Wallingford, UK: CAB International.

Berrens, R., Brookshire, D., McKee, M. & Schmidt, C. (1998) Implementing the safe minimum standard approach: Two case studies from the U.S. endangered species act. *Land Economics* **74**: 147–161.

Berrens, R., McKee, M. & Farmer, M. (1999) Incorporating distributional considerations in the safe minimum standard approach: endangered species and local impacts. *Ecological Economics* **30**: 461–474.

Berrens, R. & Polasky, S. (1995) The paretian liberal paradox and ecological economics. *Ecological Economics* **14**: 45–56.

Bishop, R. (1978) Endangered species and uncertainty: the economics of a safe minimum standard. *American Journal of Agricultural Economics* **57**: 10–18.

Bishop, R. (1980) Endangered species: an economic perspective. *Transactions of the North American Wildlife Conference* **45**: 208–218.

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

Bishop, R. & Woodward, R. (2000) Sustainability, economy and the environment. In: *Environmental Economics: Theory, Application and Policy*, ed. D. Chapman, pp. 1–42. Reading, MA, USA: Addison Wesley and Longman Publishers.

Booth, D. (1994) *Valuing Nature: The Decline and Preservation of Old-Growth Forests*. Lanham, MD, USA: Rowan and Littlefield: 287 pp.

Bromley, D. (1989) *Economic Interests and Institutions*. New York, USA: Basil Blackwell: 274 pp.

Bromley, D. (1997) Constitutional political economy: property claims in a dynamic world. *Contemporary Economic Policy* **15**: 43–54.

Brown, G. & Shogren, J. (1998) Economics of the endangered species act. *Journal of Economic Perspectives* **12**: 3–20.

Brown, G. & Szweirzbinski, J. (1988) Optimal genetic resources in the context of asymmetric public goods. In: *Environmental Resources and Applied Welfare Economics*, ed. V.K. Smith, pp. 91–118. Washington DC, USA: Resources for the Future.

Carson, R., Flores, N. & Mitchell, R. (1999) The theory and measurement of passive-use value. In: *Valuing Environmental Preferences*, ed. I. Batemen & K. Willis, pp. 97–130. Oxford, UK: Oxford University Press.

Castle, E. (1996) Pluralism and pragmatism in the pursuit of sustainable development. In: *Forestry, Economics and the Environment*, ed. W. Adamowicz, P. Boxall, M. Luckert, W. Phillips & W. White, pp. 1–9. Wallingford, UK: CAB International.

Castle, E. & Berrens, R. (1993) Economic analysis, endangered species and the safe minimum standard. *Northwest Environmental Journal* **9**: 108–130.

Castle, E., Berrens, R. & Polasky, S. (1996) The economics of sustainability. *Natural Resources Journal* **36**: 475–490.

Chapman, D. (2000) *Environmental Economics: Theory, Application and Policy*. Reading, MA, USA: Addison Wesley and Longman Publishers: 415 pp.

Ciriacy-Wantrup, S.V. (1947) Capital returns from soil conservation practices. *Journal of Farm Economics* **29**: 1181–1196.

Ciriacy-Wantrup, S.V. (1952) *Resource Conservation: Economics and Policy*. Berkeley, CA, USA: University of California Press: 395 pp.

Ciriacy-Wantrup, S.V. (1955) Benefit-cost analysis and public resource development. *Journal of Farm Economics* **37**: 676–689.

Ciriacy-Wantrup, S.V. (1961) Multiple use as a concept for water and range policy. In: Water and range resources and economic development of the West, Western Agricultural Economics Research Council, Report 9, Tucson, AZ, USA.

Costanza, R. & Daly, H. (1992) Natural capital and sustainable development. *Conservation Biology* **6**: 37–46.

Crowards, T. (1998) Safe minimum standards: costs and opportunities. *Ecological Economics* **25**: 303–314.

Crowards, T. (1999) Combining economics, ecology and philosophy: safe minimum standards of environmental protection. In: *Valuation and the Environment: Theory, Practice and Methods*, ed. M. O'Connor & C. Spash, pp. 223–240. Cheltenham, UK: Edward Elgar.

Cummings, R., Brookshire, D. & Schulze, W., eds. (1986) *Valuing Environmental Goods: An Assessment of the Contingent Valuation Method*. Totowa, NJ, USA: Rowan and Allanheld: 270 pp.

Daly, H. (1994) Operationalizing sustainable development by investing in natural capital. In: *Investing in Natural Capital*, ed. A. Jansson, M. Hammer, C. Folke & R. Costanza, pp. 22-37. Washington, DC, USA: Island Press.

Diamond, P. & Hausman, J. (1994) Contingent valuation: is some number better than no number? *Journal of Economic Perspectives* **8**: 45–64.

Dobson, A., Rodriguez, J., Roberts, W. & Wilcove, D. (1997) Geographic distribution of endangered species in the United States. *Science* **275**: 550–553.

Ecological Society of America (1995) Strengthening the use of science in achieving the goals of the Endangered Species Act. Unpublished Report, Ad Hoc Committee on Endangered Species, Washington, DC, USA.

European Commission (1999) Natural Resource Aspects of Sustainable Development in the European Union. Report of the European Commission to the 5th and 8th sessions of the United Nations Commission on Sustainable Development, Berlin [WWW document]. URL http://www.un.org/esa/agenda21/natlininfo/countr/eu/natur.html

Farmer, M. & Randall, A. (1998) The rationality of a safe minimum standard. *Land Economics* **74**: 287–302.

Folke, C., Hammer, M., Costanza, R. & Jansson, A. (1994) Investing in natural capital–why, what and how? In: *Investing in Natural Capital*, ed. A. Jansson, M. Hammer, C. Folke & R. Costanza, pp. 1–21.Washington, DC, USA: Island Press.

Foy, G. (1990) Economic sustainability and the preservation of environmental assets. *Environmental Management* **14**: 771– 778.

Goodstein, E. (1999) *Economics and the Environment*. 2nd edition New Jersey, USA: Prentice Hall: 558 pp.

Hagen, D., Vincent, J. & Welle, P. (1992) Benefits of preserving old-growth forests and the spotted owl. *Contemporary Policy Issues* **10**: 13–26.

Hamlin, A. (1989) Rights, indirect utilitarianism and contractarianism. *Economics and Philosophy* **5**: 167–187.

Hanemann, W.H. (1994) Valuing the environment through contingent valuation. *Journal of Economic Perspectives* **8**: 19–44.

Hanemann, W.H. (1995) Improving environmental policy: are markets the solution? *Contemporary Economic Policy* **13**: 74–79.

Hanley, N. (1992) Are there environmental limits to cost benefit analysis? *Environmental and Resource Economics* **2**: 33– 59.

Hausman, J., ed. (1993) *Contingent Valuation: A Critical Assessment*. New York, USA: North Holland: 503 pp.

Hohl, A. & Tisdell, C. (1993) How useful are environmental safety standards in economics? The example of safe minimum standards for the protection of species. *Biodiversity and Conservation* **2**: 168–181.

Howarth, R. & Norgaard, R. (1992) Environmental valuation under sustainable development. *American Economic Review* **82**: 473–477.

Hubin, D. (1994) The moral justification of benefit/cost analysis. *Economics and Philosophy* **10**: 169–194.

Hyde, W. (1989) Marginal costs of managing endangered species: the case of the red cockaded woodpecker. *The Journal of Agricultural Economic Research* **41**:12–19.

Innes, R., Polasky, S. & Tschirhart, J. (1998) Takings, compensation and endangered species protection on private lands. *Journal of Economic Perspectives* **12**: 35–52.

Jones, C. & Pease, K. (1997). Restoration-based compensation measures in natural resource liability statutes. *Contemporary Economic Policy* **15**: 111–122.

Kant, I. (1949) *Fundamental Principles of the Metaphysics of Morals*. (Translated by T. Abbott.) Indianapolis, IN, USA: Bobbs-Merrill: 30 pp.

Lesser, J., Dodds, D. & Zerbe, R. (1997) *Environmental Economics and Policy*. Reading, MA: Addison Wesley Longman: 751 pp.

Lesser, J. & Zerbe, R. (1995) What can economic analysis contribute to the sustainability debate? *Contemporary Economic Policy* **13**: 88–100.

Loomis, J. & White, D. (1996) Economic benefits of rare and endangered species: summary and meta-analysis. *Ecological Economics* **18**: 197–206.

Metrick, A. & Weitzman, M. (1998) Conflicts and choices in biodiversity preservation. *Journal of Economic Perspectives* **12**: 21–34.

Norton, B. (1987) *Why Preserve Natural Variety*? Princeton, USA: Princeton University Press: 281 pp.

Norton, B. (1995) Evaluating ecosystem states: two competing paradigms. *Ecological Economics* **14**: 113–128.

Norton, B. & Ulanowicz, R. (1992) Scale and biodiversity policy: a hierarchical approach. *Ambio* **21**: 246–249.

Palmini, D. (1999) Uncertainty, risk aversion and the game theoretical foundations of the safe minimum standard: a reassessment. *Ecological Economics* **29**: 463–472.

Pearce, D. (1976) The limits of cost benefit analysis as a guide to environmental policy. *Kyklos* **29**: 97–112.

Perrings, C. (1994) Biotic diversity, sustainable development, and natural capital. In: *Investing in Natural Capital: The Ecological Economics Approach to Sustainability*, ed. A. Jansson, M. Hammer, C. Folke & R. Costanza, pp. 92–112. Washington, DC, USA: Island Press.

Perrings, C. & Pearce, D. (1994) Threshold effects and incentives for the conservation of biodiversity. *Land Economics* **4**: 13–28.

Pezzey, J. (1992) Sustainability: an interdisciplinary guide. *Environmental Values* **1**: 321–362.

Polasky, S., Doremus, H. & Rettig, B. (1997) Endangered species conservation on private land. *Contemporary Economic Policy* **15**: 66–76.

Randall, A. (1985) Methodology, ideology, and the economics of policy: why resource economists disagree. *American Journal of Agricultural Economics* **67**: 1022–1029.

Randall, A. (1991) The value of biodiversity. *Ambio* **21**: 63–68.

Randall, A. & Farmer, M. (1995) Benefits, costs and the safe minimum standard of conservation. In: *The Handbook of Environmental Economics*, ed. D. Bromley, pp. 26–44. Cambridge MA, USA: Blackwell.

Ready, R. & Bishop, R. (1991) Endangered species and the safe minimum standard. *American Journal of Agricultural Economics* **73**: 309–312.

Rodgers, M. & Sinden, J. (1994) Safe minimum standard for environmental choices: old-growth forest in New South Wales. *Journal of Environmental Management* **41**: 89–99.

Rohlf, D. (1989) *The Endangered Species Act: A Guide to its Protections and Implementation*. Stanford CA, USA: Stanford Environmental Law Society: 207 pp.

Rubin, J., Helfand, G. & Loomis, J. (1991) A benefit–cost analysis of the northern spotted owl. *Journal of Forestry* **90**: 25–29.

Sagoff, M. (1988) *The Economy of the Earth*. Cambridge, MA, USA: Cambridge University Press: 271 pp.

Sen, A. (1987) *On Ethics and Economics*. New York, USA: Basil Blackwell.

Schaeffer, D. & Cox, D. (1992) Establishing ecosystem threshold criteria. In: *Ecosystem Health*, ed. R. Costanza, B. Norton & B. Haskell, pp. 127–148. Washington DC, USA: Island Press.

Scott, A. (1999) Trust law, sustainability and responsible action. *Ecological Economics* **31**: 139–154.

Shavell, S. (1993) Contingent valuation of the nonuse value of natural resources: implications for public policy and the liability system. In: *Contingent Valuation: A Critical Assessment*, ed. J. Hausman, pp. 371–388. New York, USA: North Holland.

Shogren, J. & Hayward, P. (1997) Biological effectiveness and economic impacts of the endangered species act. *Land and Water Law Review* **32**: 531–550.

Solow, A. & Polasky, S. (1994) Measuring biological diversity. *Environmental and Ecological Statistics* **1**: 95–107.

Solow, A. & Polasky, S. (1999) The endangered species act as a tool to conserve biological diversity. Choices **14**: 17–23.

Solow, R. (1992) An almost practical step toward sustainability. Published monograph, 40th Anniversary Lecture, Washington DC, USA: Resources for the Future.

Soulé, M., ed. (1987) *Viable Populations for Conservation*. Cambridge, UK: Cambridge University Press.

Stevens, T., Echeverria, J., Glass, R., Hager, T. & More, T. (1991) Measuring the existence value of wildlife: what do cvm existence values show? *Land Economics* **67**: 390–400.

Stroup, R. (1997) The economics of compensating property owners. *Contemporary Economic Policy* **15**: 55–65.

Thomas, J.W. & Verner, J. (1992) Accommodation with socio-economic factors under the endangered species act – more than meets the eye. *Transactions of the North American Wildlife and Natural Resources Conference* **57**: 627–641.

Tisdell, C. (1990) Economics and the debate about preservation of species, crop variety and genetic diversity. *Ecological Economics* **2**: 77–90.

Toman, M. (1994) Economics and sustainability: balancing trade-offs and imperatives. *Land Economics* **70**: 399–413.

Turner, R.K. (1999) The place of economic values in environmental valuation. In: *Valuing Environmental Preferences*, ed. I. Batemen & K. Willis, pp. 97–130. Oxford, UK: Oxford University Press.

Turner, R.K., Doctor, P. & Adger, N. (1994) Sea-level rise and coastal wetlands in the UK: Mitigation strategies for sustainable management. In: *Investing in Natural Capital*, ed. A. Jansson, M. Hammer, C. Folke & R. Costanza, pp. 266–290. Washington, DC, USA: Island Press.

Vatn, A. & Bromley, D. (1995) Choices without prices without apology. In: *The Handbook of Environmental Economics*, ed. D. Bromley, pp. 3–25. Cambridge MA, USA: Blackwell.

Vaughn, G. (1997) Seigfried Von Ciriacy-Wantrup and his safe minimum standard of conservation. *Choices* **12**: 30–33.

US Army Corps of Engineers (2000) Appendix I- Economics: Draft Feasibility Report and Environmental Impact Statement. Lower Snake River Juvenile Salmon Migration Feasibility Study. Walla Walla, WA, USA: US Army Corps of Engineers: 308 pp. (WWW document). http://www.nww.usace.army.mil/Isifseis/studykit/studypage.html

Walsh, R. (1992) Appendix B: Empirical evidence on benefits of protecting old growth forests and the spotted owl. In: *Economic Analysis of Critical Habitat Designation Effects for the Northern Spotted Owl*, pp. B1-B18. Washington DC, USA: US Fish and Wildlife Service.

WCED (1987) *Our Common Future*. Oxford, UK: Oxford University Press: 383 pp.

Weitzman, M. (1974) Prices versus quantities. *Review of Economics and Statistics* **41**: 447–491.

Woodward, R. & Bishop, R. (1997) How to decide when experts disagree: uncertainty-based choice rules in environmental policy. *Land Economics* **73**: 492–507.