Theoretical methodologies for Sustainability Policy: Historical development and logical structure

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ABSTRACT

This paper presents a broad taxonomy of policy methodologies that have been proposed for sustainability. Methodological development is tracked historically from a) unidirectional causation models in an "environmental impact" framework to b) dynamical models within a "natural capital" framework to c) more recent "resilience and adaptability" frameworks that will require fuller, more explicit, representation of unpredictability.

The relation between these policy frameworks is not one of mutual exclusion. Contrariwise, each successive causal modeling and policy analysis methods is better understood as a generalisation of previous ones. This clarifies the more specific contexts under which the simpler frameworks can be validly applied as special cases of the general.

Introduction

Sustainability policy, theory and practice present a diverse, and apparently confusing, array of claims and proposals. This paper presents a taxonomy of sustainability policy methodologies, allowing them to be systematically compared. In roughly historical order, sustainability has been characterised as (a) the reduction of environmental impact, (b) the sustaining of natural capital and (c) the sustaining of adaptive capacity. While each approach is motivated by distinct ethical bases they also differ significantly in the causal models used and the corresponding policy analysis methods.

This paper focuses on the latter where we see an historical development from a) comparatively simple unidirectional causation models, with extensions to represent unpredictability, to b) explicit accounts of dynamics and feedback to c) fuller, more explicit, representation of unpredictability in addition to dynamics. These model classes are not mutually exclusive, the more sophisticated classes are generalisations of the less, making clear when a simpler model class can be validly applied as a degenerate special case of a more general one. This paper briefly summarises the methodological component of a technical report by the same authors (Brinsmead and Hooker 2004), reviewing sustainability policy ethics and method.

Different sustainability frameworks also differ in their ethical bases, the characterisation of the value(s) that the policy is to sustain. However there is no necessary logical correlation between ethical basis and decision making methodology: each can be applied within a variety of ethical frameworks. The emphasis here is on decision making methodology, and so the ethics either implicit or explicit in particular applications will be mentioned only briefly.

Sustainability as a Policy Tool

The primary purpose of a sustainability concept is to inform *decisions* and *actions* aimed at concretely preserving the good, including an appropriate, long term relationship between humans and the environment, that is to answer “What ought we to do so the value in both human systems and natural environmental systems can be sustained?”

The Brundtland conception of sustainability which “meets the needs of the present without compromising the ability of future generations to meet their own needs” focuses on sustaining the ability to meet human needs, this is the good to be preserved or enhanced. Others have argued that the environment should be accorded an inherent value and hence appear explicitly. However, provided that ‘the good’ is made explicit, the general formulation of the sustainability problem above adequately characterises a wide range of positions.

However a definition of sustainability, even that which makes explicit the valuable conditions that ought to be sustained, nevertheless does not constitute a full policy methodology. In addition is required a) a causal model relating policy or action options to consequential effects and b) a method for selecting amongst those options. Since the particular policy methodology employed will constrain the range of actions pragmatically realised, and by extension the ethical values satisfied, careful attention ought to be paid to the capacities and limits of methodology.

Policy Methodology

Summarised below are a number of model classes, on which sustainability policy is based, and which are to be here compared. Each deals with change and unpredictability with differing degrees of explicitness.

Static, Unidirectional Causation Models represent cause-effect relationships as fixed and known, ignoring both structurally dynamic relationships and unpredictability (eg. Goedkoop and Spriensma 2000). The corresponding policy is usually to select those decision action causes with the greatest response coefficient. These generally suffice for implementing policies that emphasise the reduction of direct environmental impacts (poisons introduced, habitat clearing, and so forth) on numbers of creatures, size of habitat area etc.

Static Models with Uncertainty are slightly more sophisticated, and explicitly represent uncertainty in predicted outcomes, but still ignore structurally dynamic relationships. The corresponding policy is usually to ensure that some aggregate measure across all represented possibilities (such as an expected value or worst case) is appropriately optimised. Alternative possibilities include the precautionary principle or maximising a probablistic expected value.

Explicitly Dynamical Models are generalised in a different direction to that of static models with uncertainty. They are structurally dynamic, but remain predominantly determinate, at best superficially treating unpredictability. The corresponding policy is usually to ensure that some valuable ongoing services such as fresh air, clean water or economically valuable biomass production are sustained, possibly allowing trade-offs among time intervals which requires a method for making such intertemporal comparisons (eg. via a discount rate). Such analytical models are often used under resource economics and ecological economics frameworks. Natural capital sustainability policies (Pearce and Turner 1990) seek to sustain the productive capacity of natural systems, necessary for the continued production of valuable services over time.

Models Representing Unpredictability. Recently, sustainability conceptions have emphasised "resilience" and "adaptive capacity" (Folke et al. 2002), the capacity of organised ecological or economic systems to function despite externally induced changes, either shocks (brief extreme weather events) or long term alterations (land clearing and technological change). The underlying models must be explicitly structurally dynamic and represent specific classes of unpredictability. The corresponding policy is typically to ensure that valuable system functioning is preserved across a wide range of plausible contingencies, though necessarily requiring trade-offs among different contingency possibility classes, in turn requiring methods for comparing among future possible worlds (eg. via risk weighting). Adaptive sustainability policies seek to sustain adaptive capacity, necessary for continued functioning in the face of uncontrollable future change.

We investigate each class of policy methods in turn.

Static, Unidirectional Causation Model based Policies

Environmental impact reduction policies provide an archetypical example of static, unidirectional causation models. Such policies are focused on sustaining an undisturbed natural environment via impact reduction. The corresponding causal models are typically concatenated together in a unidirectional causation chain. They may include a burden production model, an impact production model and an impact evaluation model.

Environmental burden is a quantitative measure of energy or material fluxes: eg. energy use, natural resource use, material use, or pollution emissions (eg. Straub 1989). A *burden production model* relates some environmental burden to its physical cause (often some human activity), eg. a model of atmospheric NO_x, SO_x, and PM_{10} emissions as a function of coal input to a coal fired electricity generator, or the ecological footprint (Rees 2000) burden production model.

Environmental impacts are consequential changes to the natural environment due to physical burdens, selected because of their environmental and/or socio-economic value. DDT (Dichloro-Diphenyl-Trichloroethane) is always a burden but becomes an impact only when it eliminates game fish and peregrine falcons because of their ecological and/or economic value. The Life Cycle Analysis standards distinguish between burdens, which are covered by Inventory Analysis, and impacts, which are covered in Impact Assessment.

Direct physical measures of environmental burden may be only *indirectly* related to impacts, such as coral reef bleaching caused by global climate change or respiratory disease caused by NO_x, SO_x, PM_{10} emissions. The environmental burden of DDT at a farm boundary is typically much less than its concentrations in game fish where it has significant impacts. Also, impacts often occur only after substantial delays: consider the 40 year latency of mesothelioma due to asbestos inhalation. An *impact production model* relates environmental burdens to their consequent impacts.

However, burden and impact production models are not yet sufficient for determining policy. The elimination of different burdens may conflict with each other. Water recycling in coal-fired power stations decreases water usage, at the expense of increased water temperature, impairing turbine efficiency and increasing carbon dioxide emissions. The elimination of environmental burdens may conflict with valuable features of the human condition including material prosperity. Hence it is desirable to select which impacts to reduce, and by how much. This requires *impact evaluation models*, allowing the evaluation of proposed changes to proceed by the weighing up among alternative environmental impacts, and also against financial or social impacts.

Hence surrogate measures of the consequential value of environmental impacts have been developed. Some examples include the Eco-Indicator Point (Pt Goedkoop and Spriensma 2000), land area equivalent as in the ecological footprint (Rees 2000), DALYs and QALYs for human health impacts (Gold, Stevenson and Fryback 2002) or dollar equivalent (eg. ExternE). Thus among consequential value may be included not only biodiversity impacts, but also human health impacts, aesthetic considerations corresponding to waste dump ugliness and the intrusion of otherwise ‘green’ wind turbines in scenic beauty spots.

The burden production model, impact production model and impact evaluation model together comprise a full *environmental damage model*. Each, from burden, to effect, to damage, is increasingly contingent upon evaluative assumptions, and regarded as increasingly less objective. However this paper does not discuss in detail the normative implications of particular impact selection and evaluation methods. We note only that a) impact evaluation models are explicitly normative in the calculation of surrogate evaluation measures, and b) burden production and impact models are i) implicitly normative in the selection of particular impacts and burdens as sufficiently relevant to warrant explicit modeling and ii) leave specific policy assessments outside the methodology. Note also that the use of environmental impact models implies that the valuable condition to be sustained is an undisturbed natural environment. Normative criteria can be extended to human value while retaining the same class of causation models, eg including human accident and disease rates as impacts, whence the valuable condition to be sustained is an undisturbed healthy human condition.

Policy Method

The *decision making methodology* associated with unidirectional causation models is to search for system modifications, and/or policies affecting system inputs and outputs, that maximise the favourable outcomes. The typical policy recommendation consists of a discrete (once-off) design modification (e.g. capping a waste pipe) that is intended to achieve a reduced net burden or impact. Over a longer time frame a series of discrete changes might be recommended.

Limitations

While such methods have the merit of being comparatively simple, there are five types of limitation that restrict the scope of their applicability. Their frequent linearity and spatial aggregation are more obvious, and recognized and addressed, limitations while there are three more significant issues: being static, they cannot represent dynamical change over time; being causally unidirectional, they cannot represent feedback; as determinate, they cannot adequately represent situations where there is significant risk or uncertainty. We briefly discuss these limitations in order.

Limitations of Proportional Linearity and Spatial Aggregation

There is a tendency to employ linear proportional causal models for burden and impact production particularly because they are simple: the ecological footprint concept (Rees 2000) is implicitly based on an approximate model of total land requirements as directly proportional to human population. The majority of the methodological effort is then expended in the calculation of appropriate coefficients of proportionality. However there is a risk that these coefficients will be assumed to be constant across both time and space, an assumption that may or may not be realistic. Egg shell thickness in peregrine falcons may decrease roughly linearly with DDT concentration, but there is a highly non-linear threshold beyond which they break under their mother's weight destroying the young (the impact).

Similarly evaluation tradeoff coefficients are often implicitly assumed to be constant globally over both time and space. Of course they ought to be recognised as approximately constant only within some 'relevant range' (both of quantities of each impact and other contextual conditions). At extremes no further trade-offs at all will be acceptable: at some point fresh air will not be traded off any further against clean water or more production. This is well-recognised in economics, where neoclassical static models of utility are modelled as being, in general, nonlinear.

Furthermore, burden production models may not make explicit spatial variations which is required if impacts are nonlinearly density dependent. Early impact production models of chloroflourocarbon gases in the earth's atmosphere inappropriately aggregated the burden over the entire atmosphere. However, even though spatially averaged chloroflourocarbon quantities did not exceed safe levels, sufficiently high concentrations built up over just the Antarctic for the ozone layer to be depleted, a significant impact (Farman, Gardiner and Shanklin 1985).

However, it is generally well recognised that linearity is an approximating assumption and that spatial aggregation can be inappropriate (though unwarranted extrapolations are sometimes made in practice, as described above). More insidious however, are the less obvious limitations imposed by model structure, discussed following.

Limitations of Static and Unidirectional Causation Modelling

The methods advocated in the Life Cycle Analysis standards (see also Goedkoop and Spriensma 2000) implicitly employ linear (unidirectional) models of causation. This implies either that there are no significant negative or positive feedbacks, or that they are already adequately captured internally to the models. The longer the time frame under consideration, the less likely it is that unidirectional causation models will be adequate to the task of accurate policy assessment.

Newell and Wasson (2002) provide a good example of the importance of dynamical human-environment interaction, modelling human response to flooding. Nyngan is a small flood-plains community subject to random flooding events. Enhancing river levee banks appears to be sensible because it reduces flooding frequency. There is a statically optimal levee bank design that trades off construction cost against property damage prevention benefits. But the levee banks encourage economic development and fail at higher flood levels, both of which increase the costs of flood damage. Thus the dynamic feedback effects from the flood impact mitigation strategy through the regional economy falsify the foundational assumptions of that strategy.

Limitations of Predictability

Unidirectional causal models that do not represent uncertainty are invalid to the extent that the causal relationship are actually uncertain. This could potentially render policy decisions invalid, depending on how significant is that uncertainty. In sustainability policy theory these limitations are well-known, motivating the precautionary principle (Harding and Fisher 1999) and the safe minimum standards approach (Pearce and Turner 1990, pg 317) which each provide prescriptive constraints on policy in the face of unpredictability.

The precautionary principle advises that "Where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measure to prevent environmental degradation". The safe minimum standards approach (Pearce and Turner 1990, pg 317) is to "avoid irreversible environmental damage unless the social cost of doing so is unacceptably large". Application of these principles is equivalent to performing a worst case evaluation of environmental outcomes in the face of unpredictability due to respectively a) scientific ignorance and b) the possibility of wishing to reverse a decision. Both represent an arguably rational, if somewhat *ad hoc* modification to static, determinate environmental impact analysis. They thus explicitly recognise that coefficients between burden and impact, may change over time or space, or may not be known with any degree of certainty.

The examples to which the precautionary principle and safe minimum standards approaches are applied typically represent the causal model in tabular format with discrete policy options along one axis and unknown world-states along the other (eg. Neumayer 2003), giving rise to a model that is implicitly static, with unidirectional causation. While these address the limitations of a determinate analysis, they are nevertheless constrained by being static and causally unidirectional. Of course, the limitations of static, unidirectional causation models are not universally fatal, but it nevertheless remains useful to be explicitly aware of these limitations to ensure that the ensuing policy analysis is valid. Note furthermore that the application of neither policy principle is logically restricted to static, unidirectional causation models.

Conclusions

In summary, sustainability policies based on static unidirectional causation models require the modelling of the relationship between cause and effect. For a linear proportional model, this requires identification of the coefficients of proportionality. For nonlinear models an appropriate nonlinear model structure must be selected. Corresponding policy evaluation methods must weigh up among relevant consequences that are qualitatively diverse and may accrue to diverse individuals. The models are limited by an inability to represent a) change over time, b) feedback and c) unpredictability.

A good example are the models implicit in input-output analysis. Here the relationship between production factors and their products is represented as linear (constant over production scale), static (constant over time), unidirectional in causation and determinate (non-contingent). Consequently, while they are useful for a gross qualitative overview of the most significant factors (and relative importance) of production factors for any final consumption good, they are inappropriate for detailed predictions over significant time scales.

This is not to suggest that such models are useless for supporting policy. Their adequacy is to be judged, not in absolute terms of predictive accuracy, but relative to the corresponding policy decisions. An approximate model will often be policy adequate. The above analysis merely circumscribes the limits of validity of application, and suggests how the methods could be enhanced to expand their scope.

Dynamical, determinate Models

In contrast, environmental resource economic models are dynamic, because they explicitly represent the valuable system conditions that policy is intended to sustain, namely ongoing services. This marks a significant shift of normative concern away from an undisturbed natural environment as the desired condition to towards a positive, but distinctively anthropocentric, value conception, namely utility satisfaction, often the ongoing consumption of relevant services.

However, as our focus is on method rather than substantive normative issues, we simply acknowledge this, noting there is nothing logically inherent in dynamical models constraining their application to anthropogenic ends. Nonetheless, impact policy typically doesn't require detailed modelling of environmental or economic systems, just the impact production system, because if zero impact were to be achieved there would be no long term environment-economy interaction. In contrast, (ecological and economic) services management policies must consider a wider system because of the potential for feedback interactions within and between the two.

Given a unidirectional causation model of policy actions and valued consequences, any given pair of stand-alone policies (including business as usual) can be compared generating a binary decision: either the project is or is not to proceed. In contrast, for ongoing services management, the decision landscape is not well represented as independent alternatives. A decision to reduce harvesting pressure on fish stocks now will alter the future *schedules of harvest and stock combinations* possible in subsequent years.

For example, Brock, Maler and Perrings (1999) represent the dynamics of shallow lake eutrophication using differential equation models. Decisions regarding quantities of polluting phosphorous admitted to the lake are not apparent immediately, but unfold over time. The system exhibits multiple economically rational equilibria. If the lake is already in an undesirable eutrophic state, it is economically rational according to net present value maximisation to remain there, because the costs are less than the benefits of the polluting activity that maintains that state. However if the lake begins in a desirable oligotrophic state, it may again be economically rational to remain there. Thus the optimal lake state is path dependant.

In a similar dynamical vein, Perrings and Walker (1995) describe an agricultural-livestock and wildlife system subject to population dynamic constraints while Brown and Roughgarden (1995) model the population dynamics of barnacles, representing a two-stage life-cycle using continuous differential equations. The concept of maximum sustainable yield (Pearce and Turner 1990, chapter 16) of a renewable economic resource (fisheries or forestry) is based on such dynamic, but predictable, physical models.

Policy Methods

The use of dynamic causal models naturally suggests the policy method of dynamic optimisation. This requires the definition of strategy path evaluation function and selects that particular decision sequence that results in the optimal outcome. Evaluation of alternative strategy paths requires methods for comparing service flows at different points in time. A common method is to apply a discount factor to future outcomes, however there are others, such as worst-case value over the time horizon (eg Solow 1974).

The Capacities of Dynamical Models

Dynamical models allow the representation both of non-instantaneous effects and time-variation of dynamical relations. The agricultural consequences of the introduction of rabbits to the Australian continent played out over a period of decades (non-instantaneous effects). Furthermore, the specific dynamics would have been different had they been introduced fifty years later (time variation of dynamics), due to the fact that Australian agriculture would have been significantly different (additional land clearing, irrigation).

Once the time horizon is extended to any significant degree, feedback effects become potentially significant: decisions taken in the present can alter subsequent future dynamical relations. For example rabbit plagues probably actually had quite a significant effect on Australian agricultural development, in addition to Australian agricultural interests having a significant effect on rabbit populations in response. Dynamic feedback is abstracted away in unidirectional static causation models.

However, nonlinear relationships will typically be significant for all living and many complex engineering dynamical systems, in which case linear approximations of dynamic behaviour will be invalid for all but the analysis of stable behaviour close to equilibrium. The behaviour of nonlinear dynamic systems can give rise to multiple stable states (eg. Brock, Maler and Perrings 1999, Ludwig, Walker and Holling 1997), massive positive feedback effects such as reproduction and scale economies (Arthur 1990), chaotic system behavior (May 1976) or self-organisation (Collier and Hooker 1998). The longer the time horizon the more requirement there is that causal models are dynamical in order to be sufficiently accurate for policy purposes. Despite this “Most approaches to valuation...” (of policy) are usually made “...under assumptions of stability near a local equilibrium” (Folke et al. 2002, 16).

Pragmatic Limitations of Dynamical Models

The potential empirical advantages of nonlinear dynamical models is not without analytical cost. The class of model structures is much wider and the corresponding system identification techniques require complicated nonlinear optimisation methods (Sjoberg et al. 1995). Pragmatic computational considerations means in practice only highly structured dynamical models, those amenable to formal mathematical analysis, are investigated. Many earlier analyses concentrated exclusively on state-space-local dynamic stability (eg. May 1976).

Limitations of Predictability

Ludwig and Hilborn (1983) emphasise that uncertainty can invalidate both the specific conclusions, and the intuitions, from determinate dynamical analysis, making them misleadingly worse than useless. Where fisheries dynamics are known, optimal control policy recommends maintaining the population at a constant economically optimal size. The optimal size depends on parameters such as fertility, and attrition rates due to competition and predation. These parameters are unknown and must be estimated from observed data. However, the “certainty equivalent” harvesting schedule, calculated as if the best estimate of those parameters actually obtains, can result in systematic overexploitation. Ignoring the uncertainties in the dynamic parameter estimates, rather than employing a conservative strategy that is robust, or an adaptive strategy (Ludwig and Hilborn 1983) to reduce ignorance, can lead to disastrous outcomes. Hence risk considerations are fundamental to sound management.

Were there no uncertainty about future states, dynamic optimisation would suffice as a policy methodology. But uncertainty means that there is no guarantee that a strategy will bring about any specific future, let alone the service flow maximising one. In such circumstances the rational method is one of satisficing – choosing any strategy whose performance is sufficiently good – rather than optimising.

The recognition of risk can be incorporated into dynamical model based methodologies via the use of a “risk premium” component in the discount rate, or real options theory. Risk is thus incorporated via second order modifications to the basic theory. However, uncertainty may be associated with (a) the production function (Neumayer 2003); (b) the technological progress rate; (c) endogenous learning curve rates or (d) the future discount rate (Weitzmann 1998). Given the long time horizons inherent in sustainability policy, risk and *a fortiori* uncertainty are pervasive and cannot be adequately treated as merely second order theoretical considerations.

Conclusions

In summary, the development of dynamical, determinate models requires a) choice of model structure and b) the identification of parameters. Corresponding policy evaluation methods are required to weigh up or trade off among relevant consequences that i) are qualitatively diverse and may accrue to diverse individuals, and ii) occur in different time periods. They are limited by an inability to represent unpredictability/uncertainty. This being the case, it makes sense to place considerations of risk and uncertainty at the heart of the sustainability policy, bringing us to adaptiveness.

Dynamical Models with Uncertainty

Ecological analysis has always recognised the dynamical complexity characterising ecological systems such as time-lags, qualitative behavioural shifts and chaos (May 1976). Thus there is much uncertainty characterising their behaviour since neither their detailed dynamical forms nor their detailed quantitative states are known – indeed in a chaotic domain state uncertainty increases exponentially. Hence management for sustainability has increasingly focused attention on system resilience, the capacity to continue functioning despite uncontrollable change, as being of utmost importance (see Ludwig, Walker and Holling 1997 for a general theoretical account).

For example, Perrings and Walker (1995) model the dynamics of semiarid rangelands, analysing its capacity to support cattle grazing. The system state is described by the spatial distribution of plant species, their combustability, hydrological characteristics and palatability. The vulnerability of the system state to external perturbations, including extreme fire and rainfall events, varies according to both the system state and the management regime: grazing intensity and managed burning. For rangeland management, the feature of interest is the resilience of the distribution of vegetation, in particular the rangeland's capacity to graze cattle, against fire and rainfall uncertainty. If the ability of the rangeland to recover after an environmental shock decreases, then the rangeland is losing resilience (even if productivity is not affected immediately). Indeed, heavy grazing can place the system in a condition where an exogenous shock such as a fire or heavy rain storm can result in a removal of sufficient soil or grass to tip the system into a permanent decline. The endogenous dynamic response of the vegetation distribution to a range of possible exogenous perturbations including rainfall variation and fire disturbance is here significant.

Similarly, relatively recent real options developments in finance theory (Trigeorgis 1996) base their methods on causal models that represent both dynamical change and unpredictability (modelled as risk). In contrast to the discounted cash flow valuations of determinate dynamical analysis, real options valuations typically result in a premium awarded to projects with a high degree of adaptability and flexibility.

Policy Methods

Given causal models that are indeterminate, explicitly representing unpredictability, either as stochastic risk or uncertainty, the comparison of alternative strategies requires surrogate measures that effectively evaluate over a set of possible outcomes. This requirement is in addition to that of the relative evaluation over qualitatively distinct categories of outcome, and between present and future effects.

Where probabilities can be associated with uncertainties expected value methods and their variants are common. Consider the "entropy" measure of risk weighted expected value or the evaluation of a risky prospect implicit in the Capital Asset Pricing Model (Trigeorgis 1996). Conservative risk-averse evaluative analysis often recommends that a worst-case possibility should be used to define the evaluative measure (as for the precautionary principle), particularly for uncertainty that is not convertible to risk.

The policy appropriate to such models is to maximise the *ex ante* evaluation of a set of possible future outcome trajectories. This will typically involve substituting a satisficing performance criterion for optimality. The typical strategy, will then be a long term robust adaptive strategy (ideally one that meets a superior satisficing performance standard across a superior width of possible futures). This unavoidable weakening of the strategy performance criteria has the dual merit of allowing strategies to be located without requiring complete dynamical models, avoiding demands on both problem formulation and computational complexities that would attend a full analysis.

Capacities of Models with explicitly represented Unpredictability

Models can represent unpredictability regarding causal drivers, and/or the dynamical relationship between cause and effect. The first, input signal unpredictability, can be specified by a set representing all possible signal realisations. The second, dynamical model unpredictability, can be specified by a set, possibly parametrised, of dynamical functional relationships. Deep model uncertainty obtains when even the form of the dynamical relationship is completely unknown and cannot be represented explicitly in a form that enables meaningful analysis.

Models of unpredictability should be chosen to realistically represent the state of ignorance, and not chosen merely for analytical convenience. For example, “well-behaved” risk functions across possibility space include unimodal probability density functions (eg Gaussian distribution), however the actual unpredictability relationship may not be mathematically well-behaved. “Nonstandard” probability density functions such as those that are multimodal or follow inverse power laws may produce counter-intuitive (probablistic) behaviour.

Limitations of Models with Unpredictability

Together with the advantages of representing unpredictability there are significant limitations. Acknowledgement of unpredictability brings the realisation that no specified class, even a relatively broad class, of perturbation possibilities is infallible. Any attempt to explicitly model deeper classes of uncertainty - including modelling the uncertainty about uncertainty - quickly exceeds computational capacity even before degenerating into infinite regress.

Deep uncertainty is ubiquitous. Holling (1986) warns that experience with environmental management is characterised by “surprise”, the falsification of even sophisticated and complex models of environmental dynamics. Surprise occurs when the class of models used cannot in principle adequately represent the phenomenon under investigation. For example, a critical phase transition from gaseous steam to liquid water as temperature decreases is a surprise relative to the class of Maxwellian ideal gas kinematic models.

Hence Holling (1986) recommends an adaptive approach, which essentially consists of checking and rechecking whether the current system models are appropriate representations of the management problem, trying to estimate the validity *boundaries* of a particular model. The adaptive probing strategies for fisheries management suggested by Ludwig and Hillborn (1983) - in essence, the methods of adaptive control (eg Astrom and Wittenmark 1995) applied to models of ecological dynamics- is representative.

Any specified model class can be embedded within an even larger uncertainty class. Whence for any adaptive strategy, there will exist logically possible perturbations to which the system is not resilient. The essence of intelligent design is to make reasonable assumptions - narrowing of uncertainty space to a computationally tractable size - while recognising that the appropriate model class may develop as future information comes to light.

Summary

In summary, dynamical models that represent unpredictability require a choice of a) model structure, b) model parameter values, c) features to be represented as uncertain (possibly including dynamical structure), d) uncertainty representation structure and e) uncertainty parameter values. Policy evaluation methods are required to weigh up or trade off among relevant consequences that i) are qualitatively diverse and may accrue to diverse individuals, ii) occur in different time periods, and iii) occur under various contingencies. They are limited by computational tractability: both for identifying models and for their subsequent analysis. Furthermore, no nontrivial model is valid over all contingencies so that the identification of approximate boundaries of valid application are still always required.

Summary and Conclusion

We have presented various classes of causal dynamics underlying sustainability policy, and have argued that fundamental differences in policy formation and decision strategies ensue, as the corresponding analysis becomes more complex. Note that the development, from static to dynamic and from predictable to uncertain, and generally from lower to higher order, is not unique to environmental management. Similar advances have taken place in economics, management, automatic control and decision theory.

Once these classes are made explicit, the capacities and limitations of the corresponding policy methodologies become clearer. Since they are designed for their model class, the methods are useful only to the extent that the corresponding models are adequately accurate for policy purposes. Furthermore, the explicit recognition of each model class facilitates the exploration of alternative policy methodologies applicable within each class.

As the model classes become increasingly sophisticated, the conception of sustainability becomes more abstract: from the reduction of specific physical quantities, to the sustaining of productive capacity, to the preservation of adaptiveness capacity. Correspondingly, the set of physical states that satisfy the desired policy conditions is expanded and the potential decision options is richer and more open. Each conceptual transition involves a move to more openness regarding sustainability goals, and consequently more open problems which admit more uncertainty regarding relevant empirical detail, requiring more work to translate the abstract principles into policy practice.

However the last, most general, conception provides the most potential for realistic descriptions of natural and social systems. Natural systems are far too complex to know in detail and are strongly nonlinear, exhibiting self-organisational and other critical phenomena for which detailed modelling is inherently computationally intractable. Consequently sustainability policy methods must be able to cope with the implications of both nonlinear dynamics and unpredictability. The methodological tools that are based on the less sophisticated, static or determinate causal models have the merit of being simpler, and they may be validly applicable to subsystems suitably bounded in space, time and possibility. However, the more general perspective is required in order to verify that a less sophisticated model is applicable in any specific context.

Nevertheless approximation afforded by simpler models is often adequate for decision making purposes (and the computational expense of constructing more sophisticated models is nontrivial). Indeed simpler models may be valid for a majority of policy applications. However, precisely because the domain of valid applicability is so large, the corresponding methods so well developed and apparently practically successful, the capacity both to recognise when they are no longer valid and to use more sophisticated methods is arguably less than ideal.

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