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# POST-NORMAL SCIENCE AND THE ART OF NATURE CONSERVATION

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**Abstract**

Nature conservation may be considered a post-normal science in that the loss of biodiversity and increasing environmental degradation require urgent action but are characterised by uncertainty at every level. An ‘extended peer community’ with varying skills, perceptions and values are involved in decision-making and implementation of conservation, and the uncertainty involved limits the effectiveness of practice. In this paper we briefly review the key ecological, philosophical and methodological uncertainties associated with conservation, and then highlight the uncertainties and gaps present within the structure and interactions of the conservation community, and which exist mainly between researchers and practitioners, in the context of nature conservation in the UK. We end by concluding that an openly post-normal science framework for conservation, which acknowledges this uncertainty but strives to minimise it, would be a useful progression for nature conservation, and recommend ways in which knowledge transfer between researchers and practitioners can be improved to support robust decision making and conservation enactment.

**Key words:** uncertainty; researcher; practitioner; knowledge exchange; knowledge transfer; biodiversity

## Introduction

Conservation of species and ecosystems is a major global trend that has developed over the last 50 years in response to increasing evidence of environmental degradation associated with human activity (see e.g. Primack 1993; Hunter 2002). Much current global conservation policy and implementation is motivated, and guided, by the Convention on Biological Diversity, the aims of which implicitly require sound scientific understanding of the species and ecosystems that are to be conserved. The major restriction for conservation implementation is that our level of understanding of ecosystem level ecology is insufficient, because ecosystems and their components are highly dynamic, complex and unpredictable (e.g. Pahl-Wostl 1995; Kay et al. 1999; Francis, 2009). Often the direct and indirect impactors and their effects within a given ecosystem are unclear, and so it is difficult to assess which impacts most need to be addressed (Lambin et al. 2001; Srivastava, 2002; Cawardine et al., 2008). Consequently, to conserve ecosystems sustainably while still utilising their resources and services, the level of information required is beyond the current realm of scientific inquiry, let alone the capacity of governmental organisations to make reliable legislation and management decisions, and for such legalisation and management to be effectively implemented (e.g. Watzold and Schwerdtner, 2005; Papageorgiou and Vogiatzakis, 2006). Indeed, much recent scientific inquiry into ecosystem structure and processes has highlighted system complexity and the many unknowns and uncertainties that preclude prediction, management and conservation (e.g. Doak et al., 2008; Dormann et al., 2008; Francis, 2009).

Despite such uncertainties, the conservation community is driven by a supposition that something should be done immediately to preserve species and ecosystems despite our inadequate understanding of ecological principles and processes and how we may best manage and conserve them (Heywood and Iriondo, 2003). In this sense, conservation is a post-normal scientific endeavour (see Ravetz, 1999), as it represents a range of urgent problems that require immediate attention but cannot be adequately addressed by current scientific knowledge or methods, relies heavily on practitioners who are not scientific experts (an ‘extended peer community’), where decisions made may have substantial repercussions regarding human lives and livelihoods, and in which laypersons from a range of backgrounds have a stake (see Ravetz, 1999; Robertson and Hull, 2001; Rosa and Da Silva, 2005). However, the notable separation that exists between 1) the rigorous scientific investigations performed by conservation ecologists, 2) the setting of policy and legislation by governmental organisations and 3) the implementation of conservation strategies by conservation practitioners, frequently leads to a range of conservation measures being applied that often have little scientific objectivity and justification, and are consequently ineffective and can represent a waste of resources; intrinsic uncertainty is often not acknowledged (e.g. Pullin et al., 2004; Carwardine et al., 2006). With the growing governmental and public interest in ecological conservation and increasing media attention, all with expectations of success, adopting conservation methodologies that lack adequate planning or resources is arguably more harmful than beneficial.

There is also a separation between ‘biological conservation’ (a scientific discipline which has grown from the disciplines of ecology and biology and is focused on biodiversity loss) and ‘nature conservation’ (the preservation of landscapes and land use regimes originally for recreation or in the public interest, and more recently incorporating management to conserve targeted habitats or species). Although both are inseparable, as conservation of species and communities requires the conservation of biological and physical habitats, the current gap between the two approaches also confounds conservation success (see Adams, 2003). This inelegant mix of landscape management and biodiversity conservation represents broad conservation practice in many countries today. Consequently, in this paper, we use the broad term ‘conservation’ generally, to incorporate the current mix of both approaches.

Recently there have been calls for greater scrutiny of our response to biodiversity loss and conservation (see Brown and Sax, 2004; van Loon 2005; though see Larson, 2007; Peterson et al., 2007) as well as greater public inclusion in conservation (Robertson and Hull, 2001). Here, we suggest that the post-normal science concept provides a useful framework for encapsulating the totality of the conservation process, as long as 1) uncertainties at all levels are clearly acknowledged and realistic limitations are set on the reliability of decisions made, including the advantages and disadvantages of the proliferation of ‘extended facts’ and 2) all members of the extended peer community within the conservation sector are clear about their roles, and methodologies are in place to allow effective knowledge transfer between members. This paper briefly reviews the implications of ecological, ethical, philosophical and methodological uncertainties for conservation, and then discusses the post-normal conservation peer community and how it may be made more scientifically robust, using the UK conservation sector as context.

## Ecological uncertainty

Objectivity is central to the philosophy of science, though several authors have questioned whether such objectivity can ever really be achieved when considering nature and its values (Wallington and Moore, 2005; Noss, 2007; Peterson et al., 2007; Ridder, 2007). Considering our current state of ecological knowledge, there are substantial uncertainties associated with our understanding of biodiversity and ecosystems that compromise conservation efforts. Any conservation strategy other than non-interaction ideally requires an understanding of what the system consists of and how it functions to ensure that appropriate measures are taken to retain key components (e.g. Brussard et al., 1998; Hooper et al., 2005). A detailed understanding of ecosystem biodiversity and functioning is assumed and is implicit to most ecosystem management and conservation; for example Brussard et al. (1998) list ‘develop a comprehensive understanding of the ecosystem’ as one of their critical steps for ecosystem management. This is an unrealistic requirement even for ecosystems in small, well-researched and well-documented countries such as the UK.

As a recent example of an implicit acknowledgement of the level of uncertainty associated with ecological understanding, the UK Biodiversity Partnership Biodiversity Research Advisory Group (BRAG) list 163 broad research priorities that need to be addressed for more successful implementation of the UK Biodiversity Action Plan, which is the UK’s response to the Convention on Biological Diversity (Ferris, 2007). One of the research priorities within the ‘The Role of Biodiversity in Ecosystem Functioning’ theme is the need to:

> Undertake research regarding the effect of biodiversity changes or species loss on ecosystem function and the resilience and stability of ecosystem function (the difficulty of formulating a testable hypothesis needs to be recognised. It may prove impossible to answer this question, due to the many definitions of ecosystem functionality, and the large number of species in an ecosystem). Inconsistencies highlighted in previous studies require that example protocols are put forward, in order to contribute more robust evidence. (Ferris, 2007, p. 15)

This brief example demonstrates the huge uncertainties involved in our current level of understanding and the level of resources that would be required to satisfactorily address such a research gap, if indeed it can ever be addressed at a level useful for conservation.

Even a basic understanding of the many biotic components of ecosystems is often difficult to obtain (e.g. Hooper et al., 2005; Francis, 2009). A good example is our reliance on taxonomy in quantifying biodiversity (see Isaac et al., 2004). Taxonomic teaching and training has been in decline for several decades, and specialists who can identify more than one broad type of taxa are relatively rare; this situation is only now beginning to reverse, mainly because of an increasing interest in the application of new technological advances to taxonomy and an increased focus on genetics (Mallet and Willmott 2003). Whether this trend will lead to increasing abilities of scientists to accurately assess biodiversity remains to be seen. In particular, the present lack of taxonomic knowledge of underrepresented but functionally important taxa such as soil bacteria and nematodes (e.g. Gaston and Spicer 2004; Cox and Moore 2005; Lorimer, 2006), means that there is currently little accurate appreciation of biodiversity at any but the finest scales, other than for very specific groups of organisms (e.g. Cox and Moore 2005). In total we have probably only identified less than 10% of global biodiversity, and there is much uncertainty regarding species biodiversity of some taxa even in highly scrutinised ecosystems such as those in the UK (e.g. Cox and Moore, 2005; Bebber et al., 2007; see Lorimer, 2006 for a detailed and useful discussion of this in the UK context). An understanding of what constitutes whole ecosystem biodiversity, which is necessary for a robust reductionist scientific approach to conservation, is currently unobtainable. An alternative, more holistic viewpoint is that taxonomic assessment of biodiversity is not required or useful for nature conservation, as large-scale biodiversity conservation should focus on preserving functional integrity and dynamics rather than individual species and communities. Nevertheless, the current focus on species and community assessment within conservation philosophy and practice implicitly requires efforts at quantification at this level, however uncertain or potentially unattainable.

Once components have been determined, the nature of their interactions with biota and abiota must then be assessed. This includes type of interactions (e.g. predation, parasitism, infection, mutualism, competition, biogeochemical cycling) and interaction frequency, duration, and strength. This then allows some understanding of the importance of each component and its interactions in the functioning of the ecosystem. Although broad trends have been established relating higher biodiversity to more integral ecosystem processes (e.g. Loreau 2008), a paucity of information regarding functional relationships and species interactions in communities makes it difficult to assess which species or functional typologies are indicators of functional integrity, and so which should be conservation priorities (Ferris, 2007; Cabeza et al., 2008; Hendriks and Duarte, 2008).

When considering system components and their interactions as a coherent ecosystem, further uncertainty is apparent. There has been increasing acknowledgement of the complexity, non-linearity and unpredictability of ecosystems and their components over the last few decades, stemming from original investigations into complexity and variation within model ecosystems (e.g. May, 1974). There are many examples of ecosystems displaying chaos, non-linearity and threshold shifts between relatively stable states that may be brought about by relatively small changes in interactions and feedbacks (e.g. Pahl-Wostl, 1995; Rietkerk et al., 2004). This means that establishing the suitability of even the most fundamental parameters for conservation methodologies, such as the most appropriate spatial and temporal scales for implementation, and suitable target species, communities or processes, is problematic (e.g. Prendergast et al., 1999; Redford and Richter, 1999; Geneletti, 2004; Francis, 2009). Creating conservation priorities based on comprehensive scientific assessments of ecosystems is a laudable but currently unachievable ideal, particularly given the limited resources available to conduct such assessments. Such limitations should not prevent conservation from being performed—and being performed well—but should be appreciated and acknowledged within both decision-making and conservation enactment.

**Philosophical and ethical uncertainty**

Our concepts and judgements of biodiversity, the nature of evolution and our role in conservation are all large and complex topics, and our philosophical and ethical systems relating to these have emerged recently and are still relatively undeveloped (e.g. Chan et al., 2007; Lorimer, 2007; Perry and Perry, 2008; Webb and Raffaelli, 2008; Whatmore, 2002). The key concept of biodiversity is particularly problematic, and is used in a variety of scientific and political contexts. The ways in which the concept of biodiversity is framed and interpreted in relation to conservation, and the implications this has for increased uncertainty in conservation practice, is noted by Haila (1999). It is important to consider the ways in which biodiversity is being defined and its political, moral and philosophical context when making value judgements for conservation. A dichotomy of thought certainly exists in our evaluation of the intrinsic and extrinsic values of species, which is firmly based in our perception of the potential threats and benefits a species may offer us. Although we can accept that all species have a ‘right to life’ (and also a right to die; see Chessa, 2005), species certainly do not have an indefinite ‘right to exist’ (nature does not ‘conserve’ and all species are doomed to extinction at some point) and in practice conservation is not motivated by the intrinsic values of species existence. Placing value judgements on the intrinsic and extrinsic values of species is inherent in our approach to conservation (Carolan, 2008; Trudgill, 2008), yet, the issues of uncertainty should also clearly give us pause.

A useful example of dichotomy in our approach to species values can be seen—at its most pared back—in our treatment of species that are particularly harmful to humans: bacteria and viruses. From a purely objective viewpoint, these species may play, for example, important functional roles in regulating the populations of dominant species such as *Homo sapiens*; but a substantial amount of resources are invested annually in preventing the natural spread of disease-causing organisms and striving for their eradication. In 1980 the World Health Organisation triumphantly declared the two smallpox virsuses *Variola major* and *Variola minor* to be eradicated in the wild, and they have since been confined to laboratories in the USA and Russia (Slifka and Hanifin 2004). There are few situations where the eradication of a species can be so embraced by society, and of course it is entirely pragmatic that we should view these species in a different way to those which are more beneficial to our existence. In short, a spectrum of species exists, from those which are harmful to humans, those which have no perceivable benefit, those which are useful, those which provide ecosystem services, and those which are charismatic, and the range of different values we associate with these, and ambiguities in these values, create extra uncertainty in decision-making and practice (e.g. Lorimer, 2006, 2007). In particular we should be aware of the implications of value judgements, and that they are at heart self-interested.

More topically, there is currently much concern about a possible human-induced mass extinction (Diamond, 1989; Grayson et al., 2001; Lyons et al., 2004). Although there is increasing evidence of an elevated rate of extinctions due to human activity (e,g, Lande, 1998; Duffy, 2003; Michalski and Peres, 2005), the naturalness of extinction events is also of relevance for our perspectives of biodiversity and our approach to conservation. From an evolutionary perspective species extinctions, including both background and mass extinctions, are natural processes that initially have negative consequences on biodiversity, but can have long-term neutral or potentially positive impacts on global speciation and diversity, such as the increase in biodiversity following the Permian extinction event 252 ma, which may still be occurring. Uncertainty over the current and potential impacts of humans on global biodiversity and evolution raises important philosophical and ethical questions about why and how we conserve (van Loon, 2005). We can reason that human-based mass extinction, even extending over prolonged periods (e.g. thousands of years), is unlikely to significantly impact upon the actual existence of life on Earth, or the capacity of the Earth to support biodiversity and allow future speciation (though evolutionary processes may be disrupted; see Myers and Knoll [2001] but cf. Foster [1999] and O'Conner [1998]); rather, it will affect the current global environment and make it less conducive to our survival. Again, the main (implicitly acknowledged but not explicitly stated) motivating force for conservation is that our species benefits from the current environment and species assemblages; whether we are right in trying to maintain current biodiversity (which has been noted as a further potentially pragmatic anthropogenic act requiring indefinite management; see van Loon [2005]) and preventing both natural and anthropogenic species extinctions (for we cannot yet be certain of the difference) remains to be determined, and is likely to be a topic of fierce debate for many years.

**Uncertainty of anthropogenic impacts and conservation benefits**

Substantial uncertainty exists relating to the scale, strength, duration and significance of human impacts on biodiversity and ecosystem functioning, and whether we are considering such impacts at the correct spatial and temporal scales (e.g. De Leo and Levin, 1997; Spangenberg, 2007). To a certain extent, our evaluation of anthropogenic impacts and their naturalness has been limited by the historical perception of humans as being above and apart from nature, and therefore as a quasi-external influence disrupting a harmonious and balanced system (see Terrell, 2006), but of course this is far too simplistic an interpretation. *Homo sapiens* may be considered a dominant species that is out-competing many other species in a natural (or preternatural) series of processes, albeit at an unprecedented level (e.g. van Loon, 2005; Murdoch, 2001). Certainly evidence exists that humans are causing extinctions and efforts at quantifying this are becoming increasingly sophisticated (Pereira and Cooper, 2006; Brook et al., 2008), but significant uncertainty exists regarding the severity of impacts and future predictions of loss or change (e.g. Burney and Flannery, 2005; Laurance, 2007; Hui et al., 2008). Refinement of the boundaries of uncertainty and quantification of both impacts and change is essential to enable predictions of biodiversity loss as well the potential effectiveness of conservation methods, and this is likely to be a major focus of future ecological research (e.g. Thuiller et al., 2008).

With this level of uncertainty in anthropogenic impacts, the effectiveness of conservation enactment also becomes uncertain and problematic. This uncertainty is explicitly acknowledged in the 'precautionary principle' (O'Riordan and Cameron, 1994; Bräuer, 2003), which advocates some level of biodiversity conservation and impact limitation as the default option for resource and land management, due to the assumption that the effects of biodiversity loss are likely to be significant and irreversible (at least from an anthropogenic perspective). The problem with the precautionary principle is that the only effective way of ensuring that an activity does not have an unforeseen impact on biodiversity (particularly functionally important groups that are hard to quantify and not well understood) is by not performing the activity (Redford and Richter, 1999). Even simple practices performed by late-Pleistocene human groups living (by our standards) sustainable hunter-gatherer or early agrarian existences may have been responsible for biodiversity loss, particularly of mammalian megafauna (see Bulte et al., 2006; Pushkina and Raia, 2008). Some dominant species, as a characteristic of their dominance, change the environment and ambient biodiversity; this is demonstrated at a local scale by some invasive species (see Gurevitch and Padilla, 2004; Clavero and García-Berthou, 2005; Henderson et al., 2006) and is suggested as a possibility for the dominant 'dinosaur' groups before the end-Cretaceous (see van Loon, 2005). *Homo sapiens* may be no exception, albeit that the scale of impacts is far greater for our species. Furthermore, even land which is completely unused and designated as a conservation reserve may potentially be subject to detrimental external influences (e.g.

pollution, reduced immigration of species from external metapopulations, loss of genetic diversity). In practice, sustainable use of resources is required, with an accepted level of biodiversity loss and an acknowledgement of some level of unavoidable change to ecosystem functioning. Consequently our conceptualisation of conservation in the context of these anthropogenic impacts and sustainable transitions has begun to be revised to accept that some level of biodiversity loss and change is inevitable; practically, conservation in our current uncertain period of transition from industrial to sustainable socioecological regimes has to be about establishing realistic limits to biodiversity loss, and this represents an important scientific and political focus for future research.

Establishing such limits is clearly a major task and relies not just on the collection of data on species loss and impacts, but also on the collection of evidence of effectiveness of conservation methodologies, which is currently limited (see Pullin and Stewart, 2006). Pullin et al. (2004) noted that much conservation practice in the UK is informed by experience-based management and that the scientific community needs to disseminate information in the form of systematic reviews to inform decision-making. Many conservation organisations (including governmental organisations) implement projects that generate data and evidence, but do not have suitably robust designs that allow statistical analyses and testing of hypotheses and therefore do not provide the kind of peer-reviewed evidence that academically-driven conservation issues require. Frequently conservation (and restoration) projects are initiated with limited (or no) pre- and post-project monitoring, so that the effects of such activities can be judged by observation only. Furthermore, many non-academic conservation practitioners have little interest or incentive in making their results publishable; consequently many examples of both successful and unsuccessful methodologies remain unpublished or pass from practitioner to practitioner via informal networks. In this way, substantial amounts of resources and effort are being lost and/or fail to contribute to formal conservation science and improving future decision-making. In effect, the absence of evidence from the vast majority of conservation projects, which are experienced-based and conducted with little research input, often over long periods of time, represents a further source of uncertainty that needs to be addressed for future conservation practice. Without reliable knowledge of which techniques work over varying spatio-temporal scales, and which do not, progress remains painfully incremental.

**Implications of covering uncertainty with 'extended facts'**

The above discussion is intended to briefly highlight several key sources of uncertainty in conservation, and—it must be heavily stressed—is not intended to suggest that no action should be taken until the uncertainty is resolved, if it ever is; conservation is a potentially important and essential activity for the continuation of our species and civilisation, and waiting until all the evidence is in might well forestall any kind of response to biodiversity loss at all. Indeed, continual changes in our socio-ecological systems and concepts, perceptions and interactions with nature (see Castree, 2005; Fischer-Kowalski and Haberl, 2007; Forsyth, 2003) will mean that new uncertainties ('unknown unknowns') will frequently emerge (e.g. Myers, 1993; Carpenter et al., 2006). Furthermore, even with reduced uncertainty, subjectivity will remain within conservation that will affect the decision-making process, so that some mechanisms by which conservation priorities are established will remain subjective regardless of advances in our ecological understanding. Nevertheless, the implications of such uncertainty need to be acknowledged, and the uncertainty itself needs to be addressed and incorporated into our responses to biodiversity loss. In particular the placing of conservation firmly within the context of current and future developments in our ongoing socioecological transition towards sustainability (see Fischer-Kowalski and Haberl, 2007), including realism about the unavoidable loss of biodiversity and how our capacity to reduce loss may or may not change, should motivate a wide range of actors and form a good basis for decision-making. We also need to accept that society and nature change concurrently (e.g. Dickens, 2004; Castree, 2005; Fischer-Kowalski and Haberl, 2007) and that it may not be possible to entirely preserve the current environment while still allowing changes in our socioecological regime; when (and if) a sustainable society finally emerges, our local and global environments and biodiversity may well be quite different, and this should not be regarded entirely negatively.

It would also serve the conservation community well to consider how ‘extended facts’, which are non-scientific observations made by members of the extended peer community (Healy, 1999), become embedded in conservation decision-making and practice and can support progress as well as creating errors and a false sense of reduced uncertainty. Extended facts can be invaluable in the form, for example, of local knowledge held by conservation practitioners (e.g. van der Sluijs, 2007). Indeed for local-scale conservation, which is the scale at which most conservation is actually performed, such knowledge is essential for determining the necessity, potential and likely success of conservation initiatives (Healy, 1999). Negative consequences can, however, result from extended facts that may have a sound observational basis but have not been well-researched. A regional example of this is the widely held perception within the UK of *Buddleja davidii* Fran. (an introduced horticultural species) being a good plant for wildlife, particularly butterflies (hence its common name of Butterfly bush), presumably inspired by observations that Lepidoptera frequently visit the flowers of this plant (e.g. Corbet, 2000). This led to *B. davidii* being planted in many gardens and conservation areas to improve their quality as wildlife habitat at a local scale, despite long-standing evidence that the species is invasive outside its native range (China) and is a particular problem in urban areas, being frequently associated with structural damage of buildings, roads and railways (Smale, 1990; Zhang et al., 1993; Clay and Drinkall, 2001). The simple extended fact that ‘Certain butterfly species frequently visit *B. davidii* flowers and so this plant may prove to be an interesting exotic addition to garden flora’ has been reinterpreted as ‘*B. davidii* is a good plant for wildlife’, which actually contains several assumptions that introduce unacknowledged uncertainty into the use of this species in conservation practice. These are firstly that *B. davidii* attracts significantly higher numbers of butterflies than other native plants with similar characteristics, of which there is no conclusive evidence (Andersson, 2003); indeed the choice of cultivar of *B. davidii* also makes a notable difference in appeal to butterfly species (Bruner et al., 2006); and secondly that an increase in butterfly visitations significantly improves the biodiversity of the area, at least for butterflies but possibly for biodiversity in general, for which there is also no evidence. This is just a simple example, but illustrates that such ‘extended facts’ need to be carefully evaluated before being incorporated into decision-making and practice.

We now consider how the application of a post-normal science framework to conservation may help to encourage knowledge transfer between members of the extended peer community and reduce uncertainty, while encouraging an open acknowledgement that some uncertainty will always be present.

**Conservation as post-normal science and the role of the extended peer community**

The concept of post-normal science was introduced by Funtowicz and Ravetz (1993) to highlight the transition of the ‘orthodox’ practice of science from a reductionist ‘search for truth’ to the addressing of urgent problems and crises in society wherein there is substantial uncertainty and the scientific community cannot provide comprehensive answers, or solutions with an absence of risk. In the post-normal science framework, it is argued that those non-scientists who can contribute and who are willing to enter scientific dialogues should become involved in the problem-indication and -solving processes of wider societies.

From this definition, the science of conserving species and ecosystems is certainly post-normal; despite the abundance of rigorous scientific investigation that is performed by the global scientific community, most of the decisions taken that relate to conservation policy, funding and implementation are made by non-scientists while the physical implementation of conservation practice (usually at a local scale) is often performed by individuals with limited (or outdated) scientific training, often on a voluntary basis (e.g. Reidy et al., 2005; Bruyere and Rappe, 2007; Evans et al., 2008). Altogether, this represents the ‘extended peer community’ demonstrated by Ravetz (1999) that is central to post-normal science. Here, we focus on the extended peer community within the UK, firstly considering the legacy of the professionalisation of conservation.

*The professionalisation of conservation*

The separation of research and practice within the UK conservation sector may be the result of UK conservation enactment emerging from two different pathways that eventually merged due to the professsionalisation of conservation. The practice of ‘nature conservation’ emerged in the very early 20$^{th}$ century and was driven by amateurs (amateur used in the literal sense of ‘for the love’ of something) motivated by a desire to preserve, protect and enjoy wildlife and habitats within the already heavily managed UK landscape. As such, it was primarily about maintaining traditional land management practices. It was only in the 1940s that the UK government passed legislation relating to conservation, for example the establishment of the 1949 National Parks and Access to the Countryside Act, and subsequently the National Parks, which were created to allow areas for public recreation in semi-natural environments (e.g. Moss, 1999; Adams, 2003). It was acknowledged then that the setting of legislation required some understanding of the ‘nature’ that was to be conserved, if only to provide the credentials for making the policy and management decisions, and so the move towards professionalisation of conservation was begun. The subsequent establishment and development of professional governmental and non-governmental organisations with responsibility or interest in conservation, created subsequent employment opportunities, and the conservation sector gradually emerged.

In the first half of the 20$^{th}$ century, academic ecologists were mainly involved in investigating ecological theory relating to the characteristics, dynamics and diversity of ecosystems, drawing mainly on early modelling approaches and empirical fieldwork (see Golley, 1993; Benson, 2000; Francis, 2009 for reviews). The focus was on how ecosystems and species functioned, and although the preservation or conservation of these systems was implicit (generating a few publications on the issue in the 1940s and 1950s, e.g. Smith, 1947; Anon, 1959), biological conservation as a science did not emerge until the end of the 1960s, marked by the foundation of scientific journals with a specific conservation focus (e.g.

*Biological Conservation*; see Shaposhnikov, 1969). Researchers began to focus their investigations on both theoretical and applied problems in conservation, establishing how best to maintain species populations and biodiversity, and with a presumed higher level of 'scientific objectivity'. Nevertheless, although the more traditional and practice-oriented land management and broad recreation focus of nature conservation in the UK began to incorporate more research-led biological conservation, the two have developed from different principles and approaches, and have never been fully integrated (e.g. Miller, 2005). This is probably partly to blame for the current gap between the principles and methods of biological conservation and recreational land management, though both are united under the same banner of 'nature conservation'. A very general example of this is the contrasting appreciation of ecosystem dynamics within the two types of conservation. Ecosystem dynamics, variability and heterogeneity are widely acknowledged as being linked to the maintenance of species populations and biodiversity by the scientific community (e.g. Chesson and Case, 1986; Pickett et al., 1997; Morin, 1999; Ward et al., 1999), whereas traditional conservation associated with land management still perceives stability and lack of dynamism (essentially the artificial maintenance of habitats by management) as being good practice. Progress will best be made by working to explicitly combine the two approaches to conservation, which is the most logical progression though it will take much effort; or by firmly separating the two.

Higher education courses and qualifications specifically focusing on conservation followed the emergence of the scientific discipline, training people for both research and management roles within the conservation sector. These professional (and often well-qualified) conservationists emerged into a poorly-paid sector where a good professional career could generally be maintained only by becoming managers or decision-makers, and characterised by job insecurity and short-term contracts, often with little time or opportunity for research. In time it became expected that a conservation practitioner had to be sufficiently qualified, and it can be argued that this has to a certain extent displaced amateur conservationists, who remain within the volunteer workforce and align more with the land management ethos, but who nevertheless are responsible for much of the conservation practice that takes place in the UK. The saturation of the conservation sector with qualified individuals has also led to the development of ‘career volunteers’ who volunteer with conservation organisations largely in order to obtain ‘work experience’ that can help them to obtain a job in the sector, rather than due to a desire to engage in conservation enactment at a practical level.

*The UK Biodiversity Action Plan and the extended peer community*

The UK Biodiversity Action Plan (from 2003) is served by a range of organisations providing different levels of scientific and social engagement, information and action, as displayed in Figure 1. The highest level of organisation is the UK Biodiversity Standing Group, which consists of the Director of Wildlife, Countryside and Flood Management at Defra, chairs of the four UK biodiversity groups (England, Scotland, Wales and Northern Ireland), and representatives of the ‘county nature conservation agencies and the NGO community’ (JNCC, 2008). This Standing Group is advised by 1) the Biodiversity Research Advisory Group (BRAG), which advises on national and international research priorities and facilitates research in support of UK BAP objectives; and 2) the Biodiversity Reporting and Information Group, which ensures knowledge exchange throughout the UK Biodiversity Partnership structure and ensures that the action plans are appropriately maintained and reviewed. Priority habitats and species deserving of action plans are determined by the Standing Group, and responsibility for drawing up specific action plans and their enactment is devolved to the individual country biodiversity groups. These then invite conservation organisations (often but not always government funded agencies such as the Environment Agency) to lead individual action plans, with support from a range of other conservation organisations. These plans are then used as the basis for implementation of conservation measures by national, regional and local conservation groups.

This is an excellent structure for decision-making and implementation (though see Lorimer, 2006) involving an extended peer community with different expertise, knowledge and perceptions. In practice however, the notable lack of interplay between conservation research and implementation maintains many of the uncertainties discussed above. Although BRAG can highlight the key research priorities for UK conservation (see Ferris, 2007), a clear mechanism for incorporating research into the UK BAP structure (and for practitioners to feed data back into research) does not explicitly exist. It may be argued that this research is performed ‘outside’ of the system, mainly by academic researchers, but this again creates problems in terms of reliance on a small advisory group (of researchers who have full-time positions outside of this group) to filter through vast amounts of research and provide useful pointers to members of the extended community, in a top-down approach. A more effective approach would be to increase the ties between research and practice more directly, by building clearer links between organisations responsible for these elements of conservation throughout the sector. However, this is not a question of simply increasing interactions between key member organisations, but something that needs to happen across the broader ‘components’ of the sector, which can be grouped according to their priorities and functioning (Figure 2).

Within the UK, there may be considered to be three main components of the conservation sector: 1) researchers (primarily academics), who conduct short to medium term research into ecological questions that affect conservation principles and methodologies; 2) governmental (first sector) agencies (e.g. Department for Environment, Food and Rural Affairs, Environment Agency, Natural England), wherein professionals make decisions relating to conservation policy, legislation, funding allocations, training and management, and to a lesser extent, designing and implementing conservation projects; and 3) non-governmental (second and third sector) conservation organisations (e.g. Royal Society for the Protection of Birds, The Wildlife Trusts to name a couple of the larger ones; many are very small organisations with a local focus), wherein trained, semi-trained and amateur individuals enact (generally experience-based) conservation methodologies, and in some cases conduct long-term monitoring of (for example) environmental characteristics or species populations. As in the framings of post-normal science, scientifically-trained consultants play a role by advising organisations; but within conservation, this is mainly for species identification and surveying (which is often not possible by conservation practitioners), rather than for project design or assessment, and so consultants are usually in effect data providers rather than researchers or practitioners.

Despite interactions between individual organisations, there is notable separation between each broad component of the sector, characterised by limited interactions, particularly between researchers and the conservation organisations that are most frequently performing conservation in practice (e.g. see Anon, 2007; Wheeler, 2008). Figure 2 demonstrates key interactions between sector components, highlighting key methods of knowledge exchange that broadly exist, and potential methods that largely remain to be developed.

This separation has significant implications for the effectiveness of conservation in the UK, and each sector component operates according to different drivers and priorities. Although each component has its strengths, and it would be wrong to highlight only the negative aspects, nevertheless the different priorities and operational methods create problems. Ecological researchers, motivated by a desire for understanding and to publish their research, generally conduct their investigations usually over relatively short timescales (a usual three-year project or PhD funded by a UK Research Council) and then move on to a new question or topic, always keen to break new ground, and constantly chasing new funding sources. Governmental organisations are driven by an expectation that they need to legislate and act to preserve biodiversity, but this mainly consists of making and remaking decisions partly to adapt to constantly changing information (due to uncertainties and requests for decisions on best available knowledge) and also partly to demonstrate that decisions are being made and therefore matters are under control. Second and third sector conservation organisations operate under governmental legislation and advice within their means, often using experience-based management methods that have uncertain effectiveness (Pullin et al., 2004). Although some strong linkages exist between sector components and peer community members, these are generally insufficient to unite nature conservation and biological conservation adequately, and to address many conservation issues. A potential mechanism that might promote interactions include increased public-private partnerships aimed at establishing a research and evidence base for decision making in privately owned areas or in association with the management of natural resources rather than simple land preservation or experience-based conservation (e.g. Endicott, 1993). Such public-private partnerships have worked well in some areas where research and evidence collection has directly informed management, e.g. public health (Peters and Phillips, 2004), and has been noted as essential for conservation practice in some studies (e.g. Stoll-Kleemann and O'Riordan, 2002), but has so far had limited success in relation to natural resources and conservation, where long term partnerships and commitments are required (Stoll-Kleemann and O'Riordan, 2002). It is likely that some institutional change in perceptions, priorities and incentives would be needed for this to be effective.

This has implications for the effectiveness of the UK BAP, which relies heavily on a large number of actors for its implementation. To date, the UK BAP is composed of a total of 436 Habitat Action Plans and Species Action Plans, drawn from a priority list of 1149 species and 65 habitats, and relying on 1850 different organisations to enact the individual plans (JNCC, 2008). Without focusing on the strengths and weaknesses of the individual plans, the sheer number of plans and organisations involved can be interpreted in two ways: firstly, as a positive sign that a wide range of national and local organisations are involved, but secondly as an indication that the UK BAP, and the UK approach to conservation, is substantially disjointed and fragmented, and any separation or distance in knowledge exchange between the organisations involved in individual HAPs and SAPs is likely to weaken implementation (see also Pullin and Knight, 2001). At a time when there is a real political will for greater effectiveness of conservation implementation and acknowledgement and reduction of uncertainty (e.g. via evidence collation and utilisation), improving interactions between members of the extended peer community throughout all components is an essential step forwards.

We next will discuss key barriers and mechanisms for improving interactions within the extended peer community before briefly concluding.

## Developing knowledge transfer within the extended peer community

Making clear the roles of members of the 'extended peer community' of conservation practitioners within both global and local conservation sectors, and building long-term knowledge transfer links between organisations is essential to ensure that resources invested are not wasted. 'Knowledge transfer' in general is a key concern of the UK Research Councils who fund much environmental research in the UK (Anon, 2007; NERC, 2008), and there have recently been calls for academic researchers to be more involved in practice, though this has also been criticised as being unrealistic (Wheeler, 2008). In order for the conservation community to more fully acknowledge and embrace a post-normal conservation framework, each peer community member needs a wider perspective of its place within the community and the linkages that do and do not exist, as well as an understanding of the uncertainties associated with conservation. A recent workshop based at King's College London (UK), which focused on increasing knowledge transfer between Academe and practitioners within the London area, raised the question of how linkages could be improved to facilitate knowledge transfer within the peer community. The discussion identified several key barriers between researcher and practitioner community members, as follows:

1. *Differences in perceptions of the role of science within conservation.* Academics and researchers generally considered that a scientific basis to decision-making and enactment is essential for conservation, and should provide both a starting point (to justify what is being done, why and how) and an end point (to demonstrate that the enactment has an effect). Practitioners acknowledged that they would support the involvement of scientific principles and methods but considered that this was a luxury rather than a necessity, and a luxury that they could not afford in terms of time or money.

2. *Differences in academic and practitioner priorities*. Researchers generally have priorities in making a long-term contribution to the sum of scientific knowledge for conservation. This involves the acquisition of limited fixed-term research grants and the publication of peer-reviewed international articles. This is also central to their career development. Although the UK Research Councils have a requirement for knowledge transfer to encourage the application of research where possible, this remains a generally ancillary activity on the part of researchers. Practitioners primarily prioritise obtaining funds to keep people in employment in order to complete individual short-term projects. The large number of conservation organisations, often with relatively narrow local foci, also makes obtaining funding difficult. There is therefore relatively little direct incentive for knowledge transfer for either researchers or practitioners, despite both recognising its importance.

3. *Lack of understanding of respective roles*. Practitioners were largely unaware of what research was being conducted, why it was relevant to them and how the outcomes would be made available, instead responding primarily to governmental reports (which may incorporate research findings, but are often more selective summaries of findings). Researchers were generally unaware of the small, more local conservation organisations, their reason for existing and what their primary interests were.

4. *Lack of access to publications*. A key gap in knowledge transfer was the inaccessibility of scientific journals that publish research to practitioners, due to the prohibitive cost of the journals and because practitioners do not have access to institutional libraries. Although abstracts are available online, this is not sufficient for practitioners to be able to utilise publications. Likewise, a large range of data exists in the form of grey literature and informal records held by varying practitioner organisations, and which tend not to be readily noticed by researchers, who tend to rely on peer-reviewed journal articles.

5. *Uncertainties leading to inaction*. It was also noted that some of the scientific and institutional uncertainties associated with conservation (and which may result from ‘extended facts’) cause practitioners to become unwilling to invest resources in efforts that may not produce desired results. A key example of this is the uncertainty associated with climate change (what kind of change will occur and when), meaning that efforts to support habitat or species may be made redundant if ranges and habitats are changed by climatic shift. Much scientific research does lead to unclear or contradictory evidence that can lead to uncertainty in application, for example the benefits and disadvantages of planting non-native species. Evaluating such uncertainties is a key task for researchers, but this needs to be performed within a post-normal science framework.

In response to these barriers, we suggest that there are several potential mechanisms that can be implemented, certainly within the UK conservation sector, to improve interactions between the peer community, increase the scientific rigour of practice and reduce uncertainty:

1. *New means of facilitation of research findings and their implications for practice from researchers to practitioners*. These may include i) publication of summaries of methodologies or evidence for specific kinds of conservation practice, in journals or magazines that conservation practitioners have access to and are likely to read. Academic journals are frequently not of direct interest to practitioners and are expensive to subscribe to, substantially lowering their availability and appeal (partly due to academic elitism; Ravetz, 2004). A new format of journal (or potentially sections in existing heavily-subscribed popular science/nature magazines) dedicated to research summaries may help to overcome some of these boundaries, and academics should be given credit for publications in these media; indeed this is already happening to some extent with the online access journal Conservation Evidence, which has existed from 2004, although academic researchers have yet to contribute substantially to this resource; ii) allowing practitioners individual or organisational access to institutional libraries; and iii) the explicit creation of knowledge facilitator or interpreter positions (ideally funded

by governmental bodies), who may be responsible for collating, summarising and interpreting research into information and media that practitioners can effectively use. Trained in academic and research skills (e.g. via a PhD) such individuals should be able to work at varying levels within the conservation sector to ensure mutual understanding of key issues, and to help build bridges between various organisations. Note that the current suggestions of researchers working to develop knowledge transfer to bring research into practice is likely to be problematic simply because it involves a ‘stretching’ of resources and takes time and effort away from the research or the knowledge exchange; consequently, one of the two is likely to suffer. The creation of specific facilitation posts is much more likely to lead to successful knowledge transfer.

2) *Data sharing arrangements*, wherein the data from projects performed by conservation organisations is provided to researchers to analyse and publish the results so as to provide evidence and further inform conservation practice. This is most likely to be effective in situations where researchers are asked to comment on project design and implementation at an early stage, to ensure that the data will be robust and can be subjected to the appropriate statistical analysis. Such arrangements can easily be reciprocal; with a small investment of researcher time to advise such projects, a substantial amount of data could be collected to provide much needed evidence and material for publication, and is likely to be of a suitable scale and subject for wider conservation application.

3) *Collaborative research projects* supported by funding, to target applied questions. Some mechanisms for this currently exist, such as the Natural Environment Research Council Co-operative Award in Science and Engineering (NERC-CASE) studentships, but collaborative research wherein researchers utilise their experimental design and statistical skills alongside the practitioner workforce’s experience and local knowledge is also likely to be successful, and to lead to the development of lasting knowledge transfer relationships. Much more scope should be given to funding projects that utilise members from across the peer community.

4) *Mutually beneficial educational exchanges*, including researcher/practitioner workshops, practitioner organisations inviting researchers onto boards of trustees and working parties, and universities encouraging 'guest lectures' or other contributions to conservation teaching by practitioners, who may be able to provide different perspectives on the sector to academics. In particular, the creation of specific 'knowledge facilitator' posts as outlined above would allow professionals to perform many of these roles, and to dedicate an appropriate amount of time to them, avoiding the need for researchers/practitioners to perform these roles outside of their usual responsibilities and thereby devaluing the knowledge exchange.

**Conclusions and future directions**

This review has briefly summarised the ecological, ethical, philosophical and methodological uncertainties inherent in both the science and practice of nature conservation, and highlighted ways in which the extended peer community can work together to both acknowledge and reduce this uncertainty within a post-normal science framework. Fragmentation of the current UK conservation sector is due in part to the different pathways by which nature conservation as both a science and a practice developed, and several barriers exist that restrict knowledge transfer between different academic, governmental and non-governmental members of the sector. These include barriers of perception, priorities, understanding, resources and the various uncertainties discussed above. There is now a need to ensure that a more focused and cohesive conservation community emerges as we work towards a more sustainable society, and this may be achieved by increasing exchanges of people, publications, data and funding between organisations. In particular, future conservationists should acknowledge the uncertainty inherent to all levels of nature conservation, and ensure that such uncertainty is taken into account in management. These uncertainties will continue to emerge as relationships between society and nature change and it is these uncertainties that require more and better joint efforts among scientists, practitioners *and* managersin their conservation efforts.

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**References**

Adams, W. M. (2003). Future nature: A vision for conservation. 2nd ed. London: Earthscan Publications Limited.

Andersson, S. (2003). Foraging responses in the butterflies *Inachis io*, *Aglais urticae* (Nymphalidae), and *Gonepteryx rhamni* (Pieridae) to floral scents. *Chemoecology*, *13*, 1-11.

Anonymous. (1959). Nature conservation and ecological research. *Nature*, *183*, 205-206.

Anonymous. (2007). The great divide. *Nature*, *450*, 135.

Bebber, D. P., Marriott, F. H. C., Gaston, K. J., Harris, S. A., & Scotland, R. W. (2007). Predicting unknown species numbers using discovery curves. *Proceedings of the Royal Society B: Biological Sciences*, *274*, 1651-1658.

Benson, K. R. (2000). The emergence of ecology from natural history. *Endeavour*, *24*, 59-62.

Bräuer, I. (2003). Money as an indicator: to make use of economic evaluation for biodiversity conservation. *Agriculture, Ecosystems and Environment, 98*, 483–491.

Brown, J. H., & Sax, D. F. (2004). An essay on some topics concerning invasive species. *Austral Ecology*, *29*, 530-536.

Brook, B. W., Sodhi, N.J., & Bradshaw, C. J. A. (2008). Synergies among extinction drivers under global change. *Trends in Ecology and Evolution*, *23*, 453-460.

Bruner, L. L., Eakes, D. J., Keever, C. J., Baier, J. W., Whitman, C. S., Knight, P. R., & Altland, J. E. (2006). Butterfly feeding preferences for *Buddleja* selections in the landscape. *Journal of Environmental Horticulture*, *24*, 39-44.

Brussard, P.F., Reed, J.M., & Tracy, C.R. (1998). Ecosystem management: What is it really? *Landscape and Urban Planning*, *40*, 9-20.

Bruyere, B., & Rappe, S. (2007). Identifying the motivations of environmental volunteers. *Journal of Environmental Planning and Management*, *50*, 503-516.

Bulte, E., Horan, R. D., & Shogren, J. F. (2006). Megafauna extinction: A paleoeconomic theory of human overkill in the Pleistocene. *Journal of Economic Behavior & Organization*, *59*, 297–323.

Burney, D. A., & Flannery, T. F. (2005). Fifty millennia of catastrophic extinctions after human contact. *Trends in Ecology and Evolution*, *20*, 395-401.

Cabeza, M., Arponen, A., & Van Teeffelen, A. (2008). Top predators: hot or not? A call for systematic assessment of biodiversity surrogates. *Journal of Applied Ecology*, *45*, 976-980.

Carolan, M. S. (2008). The politics in environmental science: The Endangered Species Act and the Preble's mouse controversy. *Environmental Politics*, *17*, 449-465.

Carpenter, S. R., Bennett, E. M., & Peterson, G. D. (2006). Scenarios for ecosystem services: An overview. *Ecology and Society*, *11*, A29.

Carwardine, J., Wilson, K. A., Watts, M., Etter, A., Klein, C. J., & Possingham, H. P. (2008). Avoiding costly conservation mistakes: The importance of defining actions and costs in spatial priority setting. *PloS ONE*, *3*, e2586.

Castree, N. (2005). *Nature*. London: Taylor and Francis Ltd.

Chan, K. M. A., Pringle, R. M., Ranganathan, J., Boggs, C. L., Chan, Y. L., Ehrlich, P. R., Haff, P. K., Heller, N. E., Al-Krafaji, K., & Macmynowski, D. P. (2007). When agendas collide: Human welfare and biological conservation. *Conservation Biology*, *21*, 59-68.

Chessa, F. (2005). Endangered species and the right to die. *Environmental Ethics*, *27*, 23-41.

Chesson, P. L., & Case, T. J. (1986). Nonequilibrium community theories: Chance, variability, history and coexistence. In: J. Diamond & T. Case (Eds.), *Community Ecology* pp. 229-239. New York: Harper and Row.

Corbet, S. A. (2000). Butterfly nectaring flowers: butterfly morphology and flower form. *Entomologia Experimentalis et Applicata*, *96*, 289-298.

Cox, C. B. & Moore, P. D. (2005). *Biogeography: An ecological and evolutionary approach*. 7th edition. Oxford: Blackwell Science.

Clay, D. V., & Drinkall, M. J. (2001). The occurrence, ecology and control of *Buddleja davidii* in the UK. *Conference Proceedings of the British Crop Protection Council Conference on Weeds*, *1-2*, 155-160.

Clavero, M., & García-Berthou, E. (2005). Invasive species are a leading cause of animal extinctions. *Trends in Ecology and Evolution*, *20*, 110.

De Leo, G. A., & Levin, S. (1997). The multifaceted aspects of ecosystem integrity. *Conservation Ecology*, *1*, 3.

Diamond, J. M. (1989). The past, present and future of human-caused extinctions. *Philosophical Transactions of the Royal Society of London B: Biological Sciences*, *325*, 469-477.

Dickens, P. (2004). *Society and nature*. Cambridge: Polity Press.

Doak, D. F., Estes, J. A., Halpern, B. S., Jacob, U., Lindberg, D. R., Lovvorn, J., Monson, D. H., Tinker, M. T, Williams, T. M., Wootton, J. T, Carroll, I., Emmerson, M., Micheli, F., & Novak, M. (2008). Understanding and predicting ecological dynamics: are major surprises inevitable? *Ecology*, *89*, 952-961.

Dormann, C. F., Schweiger, O., Arens, P., Augenstein, I., Aviron, S., Bailey, D., Baudry, J., Billeter, R., Bugter, R., Bukacek, R., Burel, F., Cerny, M., De Cock, R., De Blust, G., DeFilippi, R., Diekotter, T., Dirksen, J., Durka, W., Edwards, P. J., Frenzel, M., Hamersky, R., Hendrickx, F., Herzog, F., Klotz, S., Koolstra, B., Lausch, A., Le Coeur, D., Liira, J., Maelfait, J. P., Opdam, P., Roubalova, M., Schermann-Legionnet, A., Schermann, N., Schmidt, T., Smulders, M. J. M., Speelmans, M., Simova, P., Verboom, J., van Wingerden, W., & Zobel, M. (2008). Prediction uncertainty of environmental change effects on temperate European biodiversity. *Ecology Letters*, *11*, 235-244.

Duffy, J. E. (2003). Biodiversity loss, trophic skew and ecosystem functioning. *Ecology Letters*, *6*, 680-687.

Endicott, E. (1993). Land Conservation through Public/Private Partnerships. Washington DC: Island Press.

Evans, S. M., Gebbels, S., & Stockill, J. M. (2008). 'Our shared responsibility': Participation in ecological projects as a means of empowering communities to contribute to coastal management processes. *Marine Pollution Bulletin*, *57*, 3-7.

Ferris, R. (2007). Research Needs for UK Biodiversity. London: Department for Environment, Food and Rural Affairs.

Fischer-Kowalski, M., & Haberl, H. (2007). *Socioecological transitions and global change: Trajectories of social metabolism and land use*. Cheltenham: Edward Elgar Publishing.

Forsyth, T. (2003), *Critical Political Ecology: The Politics of Environmental Science* (London: Routledge).

Francis, R.A. (2009). Ecosystem prediction and management. In: N. Castree, D. Demeritt, D. Liverman & B. Rhoads (Eds.), *A companion to environmental geography* pp. 421-441. London: Blackwell.

Foster, J. (1999), *The Vulnerable Planet: A Short Econmic History of the Enviornment* (New York: Monthly Review Press).

Funtowicz, S. O., & Ravetz, J. R. (1993). Science for the post-normal age. *Futures*, *25*, 739-755.

Gaston, K., & Spicer, J. (2003). *Biodiversity: An introduction*. Oxford: Blackwell Science.

Geneletti, D. (2004). A GIS-based decision support system to identify nature conservation priorities in an alpine valley. *Land Use Policy, 21*, 149-160.

Golley, F. B. (1993). *A history of the ecosystem concept in ecology*. London: Yale University Press.

Grayson, D. K., Alroy, J., Slaughter, R., Skulan, J., & Alroy, J. (2001). Did human hunting cause mass extinction? *Science, 294*, 1459-1462.

Gurevitch, J., & Padilla, D. K. (2004). Are invasive species a major cause of extinctions? *Trends in Ecology and Evolution, 19*, 470-474.

Haila, Y. (1999). Biodiversity and the divide between culture and nature. *Biodiversity and Conservation, 8*, 165-181.

Healy, S. (1999). Extended peer communities and the ascendance of post-normal politics. *Futures, 31*, 655-669.

Henderson, S., Dawson, T. P., & Whittaker, R. J. (2006). Progress in invasive plants research. *Progress in Physical Geography, 30,* 25-46.

Hendriks, I. E., & Duarte, C. M. (2008). Allocation of effort and imbalances in biodiversity research. *Journal of Experimental Marine Biology and Ecology, 360*, 15-20.

Heywood, V. H., & Iriondo, J. M. (2003). Plant conservation: old problems, new perspectives. *Biological Conservation, 113*, 321-335.

Hooper, D. U., Chapin, F. S., Ewel, J. J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J. H., Lodge, D. M., Loreau, M., Naeem, S., Schmid, B., Setala, H., Symstad, A. J., Vandermeer, J., & Wardle, D.A. (2005). Effects of biodiversity on ecosystem functioning: A consensus of current knowledge. *Ecological Monographs, 75*, 3-35.

Hui, D., Biggs, R., Scholes, R. J., & Jackson, R. B. (2008). Measuring uncertainty in estimates of biodiversity loss: The example of biodiversity intactness variance. *Biological Conservation, 141*, 1091-1094.

Hunter, M. L. (2002). *Fundamentals of conservation biology*. 2nd ed. London: Blackwell.

Isaac, N. J. B., Mallet, J., & Mace, G.M. (2004). Taxonomic inflation: its influence on macroecology and conservation. *Trends in Ecology and Evolution, 19*, 464-469.

JNCC (Joint Nature Conservation Committee), (2007). UK Biodiversity Action Plan. Available at: http://www.ukbap.org.uk/

Kay, J. J., Regier, H. A., Boyle, M., & Francis, G. (1999). An ecosystem approach for sustainability: addressing the challenge of complexity. *Futures*, *31*, 721-742.

Lambin, E. F., Turner, B. L., Geista, H. J., Agbola, S. B., Angelsen, A., Bruce, J. W., Coomes, O. T., Dirzo, R., Fischer, G., Folke, C., George, P. S., Homewood, K., Imbernon, J., Leemans, R., Li, X., Moran, E. F., Mortimore, M. Ramakrishnan, P. S., Richards, J. F., Skåness, H., Steffen, W., Stone, G. D., Svedin, U., Veldkamp, T.A. Vogel, C., & Xu, J. (2001). The causes of land-use and land-cover change: moving beyond the myths*. Global Environmental Change, 11*, 261-269.

Lande, R. (1998). Anthropogenic, ecological and genetic factors in extinction and conservation. *Population Ecology, 40*, 259-269.

Larson, B. M. H. (2007). An alien approach to invasive species: objectivity and society in invasion biology. *Biological Invasions*, *9*, 947-956.

Laurance, W. F. (2007). Have we overstated the tropical biodiversity crisis? *Trends in Ecology and Evolution, 22*, 65-70.

Loreau, M. (2008). Biodiversity and ecosystem functioning: The mystery of the deep sea. *Current Biology*, *18*, R126-R128.

Lorimer, J. (2006). What about the nematodes? Taxonomic partialities in the scope of UK biodiversity conservation. *Social and Cultural Geography*, *7*, 539-558.

Lorimer, J. (2007). Nonhuman charisma. *Environment and Planning D: Society and Space*, *25*, 911-932.

Lyons, S. K., Smith, F. A., & Brown, J. H. (2004). Of mice, mastodons and men: human-mediated extinctions on four continents. *Evolutionary Ecology Research*, *6*, 339–358.

Mallet, J., & Willmott, K. (2003). Taxonomy: Renaissance or Tower of Babel? *Trends in Ecology and Evolution*, *18*, 57-59.

May, R. M. (1974). *Stability and complexity in model ecosystems*. 2nd ed. Princeton, NJ: Princeton University Press.

Michalski, F., & Peres, C. A. (2005). Anthropogenic determinants of primate and carnivore local extinctions in a fragmented forest landscape of southern Amazonia. *Biological Conservation*, *124*, 383-396.

Miller, J. R. (2005). Biodiversity conservation and the extinction of experience. *Trends in Ecology and Evolution, 20*, 430-434.

Morin, P. J. (1999). *Community ecology*. Oxford: Blackwell Science.

Moss, B. (1999). The seventh age of freshwater conservation – a triumph of hope over experience? *Aquatic Conservation: Marine and Freshwater Ecosystems*, *9*, 639-644.

Murdoch, J. (2001), Ecologising Sociology: Actor-Network Theory, Co-Construction and the Problem of Human Exceptionalism, *Sociology-The Journal of The British Sociological Association 35*: 1, 111-133.

Myers, N. (1993). Biodiversity and the precautionary principle. *Ambio, 22,* 74-79.

Myers, N., & Knoll, A. H. (2001). The biotic crisis and the future of evolution. *Proceedings of the National Academy of Sciences, 98*, 5389-5392.

Natural Environment Research Council. (2008). Funding. Available at: http://www.nerc.ac.uk/funding/

Noss, R. F. (2007). Values are a good thing in conservation biology. *Conservation Biology*, *21*, 18-20.

O'Connor, J. (1998), *Natural Causes: Essays in Ecological Marxism* (London: Guilford).

O’Riordan, T., & Cameron, J. (1994). *Interpreting the Precautionary Principle*. London: Earthscan.

Pahl-Wostl, C. (1995). *The dynamic nature of ecosystems: Chaos and order entwined*. Chichester: John Wiley & Sons, Ltd

Papageorgiou, K., & Vogiatzakis, I. N. (2006) Nature protection in Greece: an appraisal of the factors shaping integrative conservation and policy effectiveness. *Environmental Science and Policy*, *9*, 476-486.

Pereira, H. M., & Cooper, H. D. (2006). Towards the global monitoring of biodiversity change. *Trends in Ecology and Evolution, 21*, 123-129.

Perry, D., & Perry, G. (2008). Improving interactions between animal rights groups and conservation biologists. *Conservation Biology, 22*, 27-35.

Peters, D. H., & Phillips, T. (2004). Mectizan Donation Program: evaluation of a public-private partnership. *Tropical Medicine and International Health, 9*, A4-A15.

Peterson, M. N., Riley, S. J., Busch, L., & Liu, J. (2007). Reconciling wildlife management's conflicted purpose with a land community worldview. *Journal of Wildlife Management, 71*, 2499-2506.

Pickett, S. T. A., Cadenasso, M. L., & Jones, C. G. (1997). Generation of heterogeneity by organisms: creation, maintenance and transformation. In: M. J. Hutchings, John, E. A. & Stewart, A. J. A. (Eds.), *The ecological consequences of environmental heterogeneity* pp. 33-52. Cambridge: Cambridge University Press.

Prendergast, J. R., Quinn, R. M., & Lawton, J. H. (1999). The gaps between theory and practice in selecting nature reserves. *Conservation Biology, 13*, 484-492.

Primack, R. B. (1993). *Essentials of conservation biology*. Sunderland, Mass: Sinauer Associates.

Pullin, A. S., & Knight, T. M. (2001). Effectiveness in conservation practice: Pointers from medicine and public health. *Conservation Biology, 15*, 50-54.

Pullin, A. S., Knight, T. M., Stone, D. A., & Charman, K. (2004). Do conservation managers use scientific evidence to support their decision-making? *Biological Conservation, 119*, 245-252.

Pullin, A. S., & Stewart, G. B. (2006). Guidelines for systematic review in conservation and environmental management. *Conservation Biology, 20*, 1647-1656.

Pushkina, D., & Raia, P. (2008). Human influence on distribution and extinctions of the late Pleistocene Eurasian megafauna. *Journal of Human Evolution*, 54, 769-782.

Ravetz, J. R. (1999). What is post-normal science? *Futures, 31*, 647-653.

Ravetz, J. (2004). The post-normal science of precaution. *Futures, 36*, 347-357.

Redford, K. H., & Richter, B. D. (1999). Conservation of biodiversity in a world of use. *Conservation Biology*, *13*, 1246-1256.

Reidy, M., Chevalier, W., & McDonald, T. (2005). Lane Cove National Park Bushcare volunteers: Taking stock, 10 years on. *Ecological Management and Restoration, 6*, 94-104.

Ridder, B. (2007). An exploration of the value of naturalness and wild nature. *Journal of Agricultural and Environmental Ethics, 20*, 195-213.

Rietkerk, M., Dekker, S. C., de Ruiter, P. C., & van de Koppel, J. (2004). Self-organised patchiness and catastrophic shifts in ecosystems. *Science*, *305*, 1926-1929.

Robertson, D. P., & Hull, R. B. (2001). Beyond biology: toward a more public ecology for conservation. *Conservation Biology*, *15*, 970-979.

Rosa, H. D. & Da Silva, J. M. (2005). From environmental ethics to nature conservation policy: Natura 2000 and the burden of proof. *Journal of Agricultural and Environmental Ethics*, *18*, 107-130.

Shaposhnikov, L. K. (1969). The science of nature conservation. *Biological Conservation*, *1*, 272-275.

Slifka, M. K., & Hanifin, J. M. (2004). Smallpox: The basics. *Dermatologic Clinics*, *22*, 263-274.

Smale, M. C. (1990). Ecological role of Buddeja (*Buddleja davidii*) in streambeds in Te Urewera National Park. *New Zealand Journal of Ecology*, *14*, 1-6.

Smith, G. F. H. (1947). Nature protection in Great Britain. *Nature*, *160*, 457-459.

Spangenberg, J. H. (2007). Integrated scenarios for assessing biodiversity risks. *Sustainable Development*, *15*, 343-356.

Srivastava, D. S. (2002). The role of conservation in expanding biodiversity research. *Oikos*, *98*, 351-360.

Stoll-Kleemann, S., & O'Riordan, T. (2002). From participation to partnership in biodiversity protection: Experience from Germany and South Africa. *Society and Natural Resources*, *15*, 161-177.

Terrell, J. E. (2006). Human biogeography: evidence of our place in nature. *Journal of Biogeography, 33*, 2088-2098.

Thuillera, W., Albert, C., Araújo, M. B., Berry, P. M., Cabeza, M., Guisan, A., Hickler, T., Midgley, G. F., Paterson, J., Schurr, F. M., Sykes, M. T., & Zimmermann, N. E. (2008). Predicting global change impacts on plant species' distributions: Future challenges. *Perspectives in Plant Ecology, Evolution and Systematics, 9*, 137–152.

Trudgill, S. (2008). A requiem for the British flora? Emotional biogeographies and environmental change. *Area*, *40*, 99-107.

Van der Sluijs, J. P. (2007). Uncertainty and precaution in environmental management: Insights from the UPEM conference. *Environmental Modelling and Software*, *22*, 590-598.

Van Loon, A. J. (2005). The dubious role of man in a questionable mass extinction. *Earth Surface Reviews*, *62*, 177-186.

Wallington, T. J., & Moore, S. A. (2005). Ecology, values, and objectivity: Advancing the debate. *BioScience*, *55*, 873-878.

Ward, J. V., Tockner, K., & Schiemer, F. (1999). Biodiversity of floodplain river ecosystems: ecotones and connectivity. *Regulated Rivers: Research and Management*, *15*, 125-139.

Watzold, F., & Schwerdtner, K. (2005). Why be wasteful when preserving a valuable resource? A review article on the cost-effectiveness of European biodiversity conservation policy. *Biological Conservation*, *123*, 327-338.

Webb, T. J., & Raffaelli, D. (2008). Conversations in conservation: revealing and dealing with language differences in environmental conflicts. *Journal of Applied Ecology*, *45*, 1198-1204.

Whatmore, S. (2002), *Hybrid Geographies: Natures, Cultures, Spaces* (London: Sage).

Wheeler, P. (2008). Government subsidized by academia on conservation. *Nature*, *451*, 1050.

Zhang, X., Xi, Y., Zhou, W., & Kay, M. (1993). *Cleopus japonicus*, a potential biocontrol agent for *Buddleja davidii* in New Zealand. *New Zealand Journal of Forestry Science*, *23*, 78-83.

**Figure Captions**

Figure 1: Structure of the UK Biodiversity Partnership and Biodiversity Action Plan implementation. UK Biodiversity Partnership members are denoted by boxes, while key processes are shown by connecting arrows.

Figure 2: Conservation sector components, with established types of knowledge transfer highlighted (solid arrows) alongside underdeveloped interactions (dashed arrows).