Ecosystem health demystified ©

An ecological concept determined by economic means

By E. J. S. Hearnshaw[1], R. Cullen[1] and K. F. D. Hughey[2]
[1]Commerce Division, Lincoln University, [2]Environment, Society and Design Division, Lincoln University

Abstract

This paper addresses "ecosystem health", a concept recently popularised as the way forward in evaluating nature. The concept is often defined in vague expressions and is being seen more as a broad societal aspiration rather than a specific performance measure of ecosystem management. As such, the paper aims to demystify ecosystem health, that is, to demarcate an accurate and feasible characterisation of the concept. To achieve this aim an examination of the various viewpoints of nature is undertaken. Models of ecosystem health, such as the notions of naturalness, genetic fitness, climax, diversity, stability and keystone species are each considered and subsequently deemed inappropriate, especially when viewing ecosystems as "complex self-organising systems". Complex self-organising systems are non-linear dynamic systems that have multiple steady states and have emergent and chaotic properties. One model that captures this self-organisation process is Holling's adaptive cycle. However, when investigating this model it was concluded that there is no means to determining which phase within a system state, or state within a system is ecologically "better". Therefore, ecosystem health cannot be considered in a positive manner established by scientific objectivity. Rather, the concept must be determined in a normative fashion through it is suggested the elicitation of subjective societal values, so to define an optimal management strategy. But, implementing such a strategy is difficult because the changing nature and unpredictability of complex self-organising systems means we cannot focus on "locking-in" ecosystems (or preferences), instead it is argued we must forever adapt to changing ecological conditions.

Keywords: Complexity, diversity, ecosystem health, naturalness, resilience, self-organisation, societal values and stability

1.0 Introduction

Ecosystem health, a concept first generally enunciated by the pioneering ecologist Aldo Leopold in 1939 as a means to elucidating the condition of ecosystems, is today being hailed as the way forward in evaluating nature and its management for conservation and resource use purposes. This interest in the concept of ecosystem health has been brought about, in part, because the management of ecosystems captures a broader, all-encompassing perspective of nature (Angermeier & Karr, 1994), which is likely to take advantage of associated gains in economies of scale and efficiency (Simberloff, 1997). Moreover, the popularity of ecosystem health also stems from an easy comprehension of the concept, as it fundamentally draws on the expressions developed in human health and makes that seemingly effortless and intuitive step to integrate these notions with ecological theory (Rapport, 1989; Costanza, 1992). Thus, one might say that ecosystem health effectively applies the human health metaphor to ecosystems (Schaeffer *et al.*, 1988). But, what makes the concept of ecosystem health so appealing is that while it focuses on ecosystems as its unit of measurement, and thus can encapsulate an ecocentric perspective, it not only draws upon human health as an analogy, but also encompasses human (societal) health as well. Indeed, even though well removed and hidden from many actors of society, a healthy ecosystem in economic terms provides the necessary "factors of production" needed for human health and basic life-support required for development (Folke, 1999). Consequently, at the root of ecosystem health, is the ideals of co-evolutionary development (Norgaard, 1984; 1994); that is, the conditions necessary to sustain the capacity of an ecosystem are very much dependent on society, and yet in turn, society is dependant on these very ecosystems for their own health and development (Costanza, 1992; Odum, 1993; Ferraro & Simpson, 2002).

Nonetheless, despite the popularity of the term ecosystem health, the concept still remains poorly understood which has resulted in it being defined, at best, utilising rather vague expressions (O'Laughlin *et al.*, 1994). In contrast, while human health has a wide body of reference data on the so-called "standard human" (Schaeffer *et al.*, 1988), advocates of ecosystem health have much less agreement in demarcating a benchmark or apt endpoint. However, difficulty in demarcating ecosystem health in precise language is not surprising. The inherently intricate nature of ecosystems coupled with certain philosophical misgivings, such as, whether anyone has ever seen an ecosystem, and is health a relevant expression when applied to a system, have plagued progress towards a suitable working definition (Calow, 1992; Costanza, 1992; Callicott, 1995; Meyer, 1997; Kapustka & Landis, 1998; Rapport *et al.*, 1998; Callicott *et al.*, 1999; Karr, 1999). What is more, the literature is besieged with confusion as to whether ecosystem health and ecosystem integrity are one and the same thing or not (*e.g.* Karr, 1996). Thus, with these reservations, simply seeking to achieve a "healthy ecosystem" provides no more direction to the ecosystem manager, than seeking a "strong economy" does for the economist (Reid, 1994).

These difficulties in defining ecosystem health have unfortunately stalled the complete endorsement of the concept (Steedman, 1994; Scrimgeour & Wicklum, 1996; De Leo & Levin, 1997; Gaudet *et al.*, 1997; Lancaster, 2000). As such, the importance of ecosystem health has been marginalised, whereby it is being portrayed more and more as a broad societal aspiration than a specific measure of ecosystem management. Thus, if ecosystem health is to be used as a means to examine the performance of ecosystems, then coherent, clear and quantifiable definitions must be crafted so that the concept can be delineated operational. That aim is what this paper endeavours to achieve, that is to say, to demystify ecosystem health, to demarcate an accurate and feasible characterisation of what a healthy ecosystem is deemed to be. In other words, what we are attempting is to delineate the appropriate performance criteria for determining the "success" of ecosystem management. In order to achieve this aim an exhaustive examination of the various viewpoints of nature is undertaken.

2.0 The arcadian naturalness model

One viewpoint of the nature of ecosystems is the arcadian school, which promotes the characterisation of ecosystems through notions of natural aesthetics and ethics towards species (Worster, 1979). Consequently, this school of thought proposes that the health of an ecosystem is determined by its degree of natural integrity (Schaeffer *et al.*, 1988; Anderson, 1991; Angermeier & Karr, 1994; Wicklum & Davies, 1995). A high degree of natural integrity is considered an ecosystem which displays considerable "naturalness", that is succinctly, an ecosystem that is unimpaired from human action and on all accounts would be considered pristine (Karr & Chu, 1999). Consequently, this so-called naturalness model of ecosystem health implicitly infers that ecosystems that are unaltered by human activity are "healthier" than human-altered systems and that ecosystem health can be described solely through the maintenance of a natural assemblage of species.

There are a number of examples of the naturalness model that have been proposed or utilised for the evaluation of ecosystem health. One, pioneering example is that given by Leopold (1941) who claimed that "wilderness" should be considered the best "base-datum of normality". Another more developed framework is the Index of Biological Integrity (IBI) (Karr, 1981; Karr *et al.*, 1986). This simple index develops a scale of health, whereby an ecosystem, which is the product of evolutionary processes in the absence of modern human activity, is considered the index's optimum. More recently, Stephens *et al.* (2002) developed a sophisticated framework to evaluate health of ecosystems by formally integrating naturalness, described as natural character, with a priority setting function reflecting the significance of the ecosystem for nature heritage and conservation purposes (see *Equation 1*).

$$\text{NH payoff} = \sum_{0}^{T=50} \frac{(\Box \text{NC}_i)}{(1+p)^t} \times [\text{DIS}_i, \text{IMP}_i, \text{SIZE}_i]$$

Equation 1: Nature heritage framework (*Source*: Adapted from Stephens *et al*., 2002).

Where NH is the nature heritage index of ecosystem *i*

ΔNC_i is the change in natural character of ecosystem *i* after ecosystem management

Priority setting function: DIS_i is the distinctiveness of the ecosystem *i*

IMP_i is the importance of ecosystem *i*

SIZE_i is the size of ecosystem *i*

Importantly, though rarely stated explicitly in the literature, the premise of the naturalness model is the ideology of environmental therapeutic nihilism, in that "nature knows best" and is subsequently "good" in itself (Commoner, 1971). As such, the naturalness model enshrines a dichotomy between humans and nature, a metaphysical separation between the "vices" of humans from the "virtues" of nature. Therefore, this ideology rejects that society is "in charge" of nature. The implications of this premise is that while systems altered by humans, may be desirable for strictly utilitarian reasons, they cannot provide a truly objective ecocentric point of reference for assessing ecosystem health (Angermeier & Karr, 1994).

The exact objective point of reference presumed by the naturalness model has nonetheless been questioned, not least because the very act of delineating what is the natural state of an ecosystem must be concluded as an implicit subjective preference (Lele & Norgaard, 1996). For example, in the case of New Zealand (or Australia), is naturalness calculated from the time of initial human arrival or at the time of European arrival? The selection as to which of these two reference points is the most appropriate according to the naturalness model is ultimately a subjective value judgement (Wassenaar & Ferreira, 2002). Indeed, even if a pre-human or a pre-European state could be universally agreed upon as a definition of what naturalness really is, there still remains the problem that in either case, our understanding of what pre-European ecosystems let alone a pre-human ecosystems once looked like is poorly known (Caughley, 1989; Pimm, 1991). As such, most attempts to implement the naturalness model are based on the earliest recorded form, which is for all intent and purposes well after European colonisation (Hannon, 1992), and even then most ecosystems can only be characterised by some components of macro-vegetation (McGlone, 1983; 1989).

For argument sake let us suppose that sufficient information could be gathered necessary to recreate the composition of an historic ecosystem. We are still however left with a number of issues that inhibit the practicality of the naturalness model. Not least of these difficulties, is that the concept of naturalness signifies the modelling of an ecosystem from a "snapshot" perspective, which neglects changes from "natural" perturbations that undoubtedly occur. Furthermore, utilising a naturalness reference point characterises an ecosystem as a eternally static system, whereby all human perturbations and other stresses outside a natural range represent a decrease in health is similarly not accounting for the fact that ecosystems evolve over time. Thus, because ecosystems are dynamic systems, even if the species are not extinct, ecosystem managers in practice would never be able to attain the "correct" assembly of species for any reasonable length of time.

A further deficiency with the naturalness model is that it only can define an ecosystem as a biotic structural assemblage of species, which seemingly ignores the functionality of an ecosystem in favour of aesthetic appearance alone (Ehrlich & Ehrlich, 1992; Thompson, 2000). Ecosystem functionality however, is important because it models and encapsulates the nature of an ecosystem truly as a system, consisting of networks, connections, mechanisms, flows and fluxes (Odum, 1969; O'Neill *et al*., 1986). Interestingly, some researchers suggest that the monitoring of ecosystem structure and composition will adequately encapsulate ecosystem functioning because functional components will most probably be maintained if the structural assembly is adequately preserved (Ferriera & Towns, 2001). To date, however, this assumption has no conclusive empirical evidence in which to support its claims.

Given the lack of scientific evidence in support of the notion that an ecosystem's structural assemblage begets its function, most restoration efforts of ecosystems have opted to avoid the attainment of a "natural" assemblage of species. Instead, because functionality may well be more significant to the health of an ecosystem, restoration programmes have often adopted functionality as the sole measure of ecosystem health. Accordingly, restoration is then considered a "success" when ecological functioning is restored to "natural" levels (Hobbs & Norton, 1996). Such a means to evaluating ecosystem functioning has been developed, in which the measures utilise a number of vital ecosystem parameters of functionality (*e.g.* nutrient cycling and retention) (*e.g.* Swanson *et al.*, 1993; Caraher & Knapp, 1995; Walker & Boyer, 1995) (see *Figure 1*).
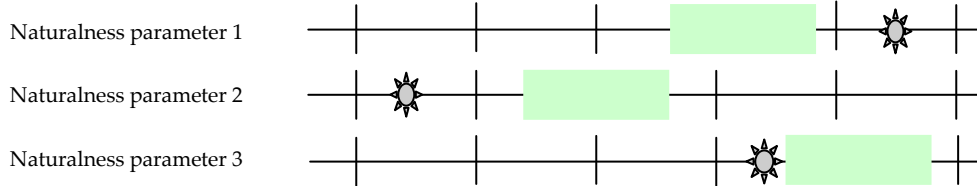

Figure 1: An example of ecosystem health evaluated based on the vital ecosystem parameters of ecosystem functioning. This approach determines the health of the ecosystem by comparing the present state of the parameter of ecosystem functioning with its delineated relative range of natural variability (*Source*: Caraher & Knapp, 1995; Hobbs & Norton, 1996).

Where ✶ is the present state of the parameter of ecosystem functioning

is the relative range of natural variability of the ecosystem

However, while this ecosystem parameter approach may be considered useful, it can be criticised for two reasons. First, it is well known that ecosystem functions, such as nutrient retention are extraordinarily difficult to quantify (Burley, 1988; Noss, 1990; Nunes *et al.*, 2003). Secondly, the restoration of ecosystem functioning is often best achieved through the addition of non-native species (Aronson *et al.*, 1993; Lockwood & Pimm, 2001). That is, non-native species substitutions are possible, which replicate the functioning of a natural ecosystem. The implications of this finding would seem to be that there is no scientific basis for "natural" pristine systems to be considered healthier than other human-modified ecosystem states and that the maintenance of structure and function of an ecosystem are indeed quite distinct projects. Recently, Loucks (2000) has attempted to integrate structure and function together so as to measure ecosystem health according to the naturalness model (see *Figure 2*).
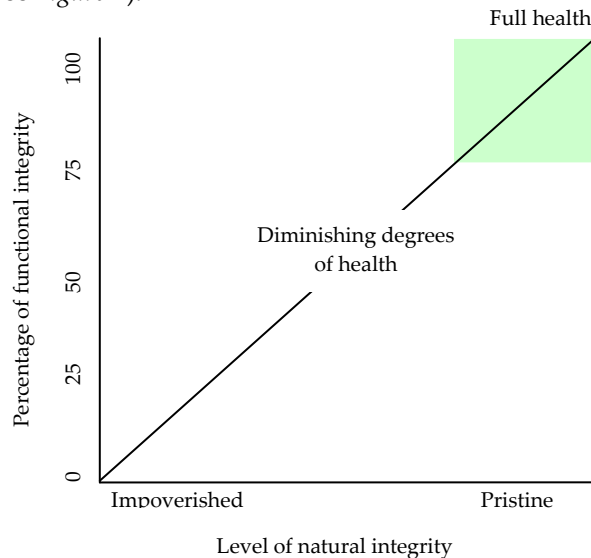

Figure 2: The continuum of functional integrity against natural (structural) integrity
(*Source*: Loucks, 2000).

Despite, the measure of ecosystem health developed by Loucks (2000), one still feels uneasy about the merits of the naturalness model, given that non-native species can improve seemingly the health of the system. However, regardless of this contradiction the greatest difficulty with the naturalness model is that paradoxically we end up finding that nature must be actively managed and restored if it is to remain fit for native species, despite the insistence of the naturalness model that requires the separation of humans from nature (Swanson, 1995; Callicott, 1998). Accordingly, if we assume that it is right when it tends to disturb the biotic assembly only at natural spatial and temporal scales, and it is wrong when it tends otherwise, it would suggest then that intensive restoration projects are impermissible insofar as they would disturb ecosystems beyond natural perturbation scales (Callicott, 1996). It would seem illogical that intuitions about the positive value of ecosystem restoration result in objecting to the naturalness model (Fairbrother, 1998). After all, "how can anything be restored by human agency the essence of which is to be independent of human agency?" (Attfield, 1994; *p*. 45).

It seems that the conception of naturalness is a misnomer. It is incorrect to assume as with the naturalness model that humans are "apart from" and not "part of" nature. Since Charles Darwin and his classic writings "The Origin of Species" (1857) and "The Descent of Man" (1871), we have established that humans are nothing more or less than a primate with some exceptional talents. We, as a society cannot afford the illusion that maintains that humans are separate from nature, as it denies the fundamental reality that humans through the intervention of development now effect to some degree all accessible ecosystems on this planet (Vitousek *et al*., 1997). Thus, it is paramount that when managing ecosystems that we acknowledge that anthropogenic changes of ecosystems are as "natural" as any other. We must come to terms with the fact that we as a society are intimately interconnected with nature, and that socio-economic systems are embedded in ecosystems (Kay & Regier, 2000; Limburg *et al*., 2002).

3.0 Imperialism, reductionism and systems theory

The shortcomings of the arcadian school and its naturalness model as a viewpoint of nature, leaves us with the other predominant school of thought that of imperialism, which is a viewpoint of nature based above all on logic and scientific analysis (Worcester, 1979). As such, the resource and functional aspects of ecosystems are prioritised, well above ethical or aesthetic considerations. However, within the imperial school of thought, an intellectual divide has separated scholars' view of nature, which at its extremes could be considered reductionistic, while at the other holistic. The reductionistic view, which is the cornerstone of conventional scientific thinking, considers that nature can be completely described through its presumed micro-foundations and the collection of its component parts. On the other hand, those of a more "holistic" persuasion insist that the essence of nature itself is best described by investigating the whole utilising the theories of systems thinking.

Needless to say, reductionists have looked to find the fundamental "building blocks" of nature. Thus, with the scientific discovery of genes, cells and molecules, life scientists, have been won over that reductionism through molecular science represent the best means of determining the underlying truths of nature. But, by embracing molecular science, modern biology has all but lost any true identification with the science of the whole. Moreover, the incorporation of Mendelian genetics with Darwinian theories of evolution, has led biologists to infer that every point in space is realisable as an organism, species and ultimately an ecosystem, as long as the localised environmental conditions favour its expression. In other words, any kind of biological and ecological form is seemingly possible, within certain basic mechanical limits. As such, when these so-called neo-Darwinists wish to speak of a healthy ecosystem, what they mean is that the conception of ecosystem health should be one based prominently on genetic diversity (or distinctiveness), as it is this which dictates the processes of natural selection, that blind, cumulative and non-directional force of nature, to continue, and thus ensures prolonged evolutionary potential. That is, genetic diversity ensures that

organisms, species and ecosystems can maintain their "biological fitness" and therefore usefulness for survival in their current and future environments.

Despite the optimism of biologists, the presumptions of neo-Darwinism theories of evolution leaves little explanation for the actual generation of highly complex ecological and biological forms found in nature. In the neo-Darwinian view, species and ecosystems are just products of adaptation somehow brought together through random genetic mutations. Thus, neo-Darwinian theories seem to have trouble providing a convincing explanation of why "higher" species ever emerged when bacteria, a simple organism, has a high degree of biological "fitness" (Davies, 1987). Extraordinarily, even Darwin was concerned with the extensive ecological and biological form that many "higher" species have. For example, in Origin of Species (1857; *p*. 172) Darwin wrote: "To suppose that the eye with all its inimitable contrivances for adjusting the focus to different distances, for admitting different amounts of light and for the correction of spherical and chromatic aberration, could have been formed by natural selection, seems, I freely admit, absurd in the highest degree possible". Moreover, with neo-Darwinism causality is neither mechanical nor deterministic, because of the role of random genetic mutations. Thus, neo-Darwinism seems to leave out any probable explanation for the intentional ecosystem development and behaviour observed in nature.

The inability of neo-Darwinism and natural selection to explain appropriately the development and behaviour of complex biological and ecological forms found in nature, has propelled the systems perspective forward, which above all claims that it is impossible to understand the nature of ecosystems by only investigating its component parts (Simberloff, 1997). The reductionistic approaches to science, to which many scientists adhere, do not seem to be met with ecosystems (Costanza, 1993). Indeed, ecologists and other system theorists contend that a system is not characterised by weak, linear interactions between component parts, but instead by strong, usually non-linear interactions between all of its parts (Costanza, 1992; Kay & Regier, 2000). This means that is it impossible to simply sum scale behaviour to arrive at large scale results (von Bertalanffy, 1968; Rastetter *et al*., 1992). Thus, dynamic systems are considered fundamentally irreducible and indivisible units (Wolfgram, 1984; Bohm *et al*., 1997). In view of that, system analysts attempt to reveal properties of ecosystems by studying the systems as a whole. It is thought that by adopting a systems perspective of the whole, certain properties become apparent and other behaviours are made detectable that otherwise would be "invisible" by processes of reductionism (Ulanowicz, 1986). Clearly then, what we need is "to see the forest, not the trees" (Jorgensen, 1997; *p*. 7).

3.1 The succession-to-climax model
One systems perspective developed in ecology is the "process-functionalist" approach (Allen & Hoekstra, 1992), which treats ecosystems, that is, organisms, species and their physical environments as "integral bio-geo-chemical energetic systems" (Lotka, 1925). Thus, the process-functionalist approach propounds that the units of selection in ecosystems are cycles of energy and material flow that have different auto-catalytic properties (Odum, 1969; Depew & Weber, 1996). Given this perspective of the underlying mechanisms of ecosystems, Lotka (1925) formulated the "maximum power principle". In essence, this principle argues that organisms tend to be selected by these energetic systems (ecosystems) when they are more efficient in their energy utilisation, and species are favoured, if they enter into cooperative interactions with other species, allowing the ecosystem as a whole to maximise the through flow of "useful" energy. Consequently, with the appropriate selection of organisms and species, the whole system is likely to increase its energy flows and material cycle rates (metabolism) through the system, which it is argued will increase total system biomass and the overall health of the ecosystem.

The behaviour and development of an ecosystem of this process-functionalist approach has been modelled according to the theory of ecological succession, a theory which ultimately leads to a stable climax phase (Odum, 1969). Accordingly, the aptly named succession-to-climax model presupposes that a highly ordered

successional sequence of biotic communities dominated by small fast growing species referred to as *r*-strategists, dynamically converge in a linear manner towards a sustained and often predictable climax assemblage of *K*-strategist species, species that are large but slow growing (Clements, 1916; Tansley, 1920; Odum, 1969). Interestingly, the notion of a mechanical Newtonian-like development towards a stable equilibrium purported in the succession-to-climax model is analogous to theories and axioms found in much of neoclassical economic theory (Kamien & Schwartz, 1991; Varian, 1992; Holling & Sanderson, 1996). Certainly, both make the assumption of relatively steady growth, with stabilising forces providing an "invisible hand" that guides the system along a trajectory to maintain productivity and towards a sustained single steady state or equilibrium. Hence, in presuming the existence of a single stable equilibrium point, the climax phase, the succession-to-climax model presumes implicitly the property of global stability (see *Figure 3*).

(a) x_t $x_t = x_{t-1}$ $x_t = f(x_{t-1})$ 45° x_1 x_{t-1}
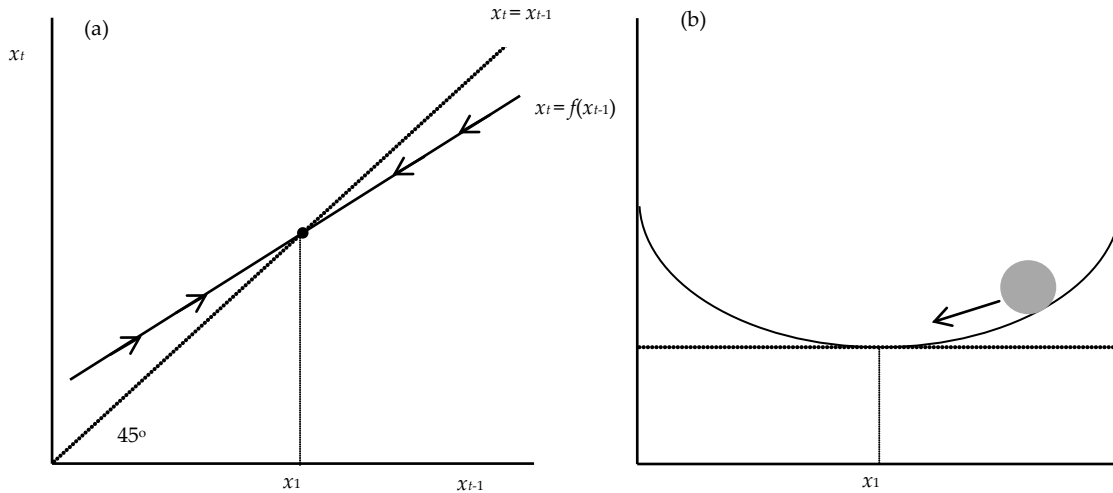
(b) x_1

Figure 3: (a) Graphical representation in discrete time of a dynamic linear model, as used in the succession-to-climax model to describe the development and behaviour of an ecosystem. (b) So-called ball-and-cup diagram (see DeAngelis & Waterhouse, 1987) illustrating that with a linear model, the system (ball) converges towards a single equilibrium x_1 (also the climax phase) no matter where the starting point.

We know however, that ecological disturbances will inevitably occur, and these perturbations prevent the development of an ecosystem attaining its steady state. However, the succession-to-climax model presumes that a disturbance will only take an ecosystem back to a previous successional phase, whereby the development towards the climax phase continues to persist predictably again after the disturbance. Thus, according to the model an ecosystem will always develop towards its climax phase regardless of how far it is displaced from this phase after a disturbance event. As such, the health of an ecosystem according to the succession-to-climax model can be modelled as if it were a homeostatic system, because homeostasis assumes fundamentally that ecosystems exhibit a unique equilibrium, which can self-perpetuate generation after generation by negative feedback (Odum, 1969; Oechel *et al.*, 1994; Kay & Regier, 2000). Importantly, a homeostatic system has intuitive appeal when defined in terms of ecosystem health as it interprets health in a similar manner to the health of humans, in that optimal ecosystem health can simply be defined as a system absent of "disease", whereby "disease" is considered a disturbance to the system (Schaeffer *et al.*, 1988; Anderson, 1991). Hence, for an ecosystem to retain its health the ecosystem "attempts" to eliminate the disease and return to its "preferred" healthy "state", the climax phase, which is absence of "disease".

From a scientific perspective, the optimal health of the system found at the climax phase can be explained by two variables: potential and connectedness. The potential of the system represents the accumulated biomass

or ecological capital developed through successional dynamics, and is therefore greatest at the climax phase (Carpenter *et al.*, 1999). The connectedness of the system determines the strength of internal connections between species that mediate and regulate the influences between internal processes and the external environment. In other words, connectedness underpins the degree of internal control that a system can exert over external variability (Ulanowicz, 1986; Holling & Gunderson, 2002), which again is at its greatest levels at the climax phase. Thus, it is not surprising that the climax phase is most likely to be made up predominantly of *K*-strategist species, which are species that have smaller specific metabolic rates compared to *r*-strategists species, which are typically found in the earlier successional phases of ecosystem development. Accordingly, one might postulate that these climax species seem to have been selected as they utilise energy more efficiently and thus, require less maintenance. One might then conclude that the *K*-strategist species are "better" adapted than are *r*-strategist species. Thus, as theorised the changes in the successional communities are made in an effort to improve the whole system's adaptation to utilise resources more effectively. Interestingly, Hannon (1992; 1999) taking the climax phase of ecosystem development as the definitive goal function of an ecosystem, has proposed a means to measuring ecosystem health termed gross ecosystem product (GEP), which is based directly on the familiar measure of economic flows gross domestic product (GDP). These explanations of ecosystem health in the succession-to-climax model are illustrated in *Figure 4*.
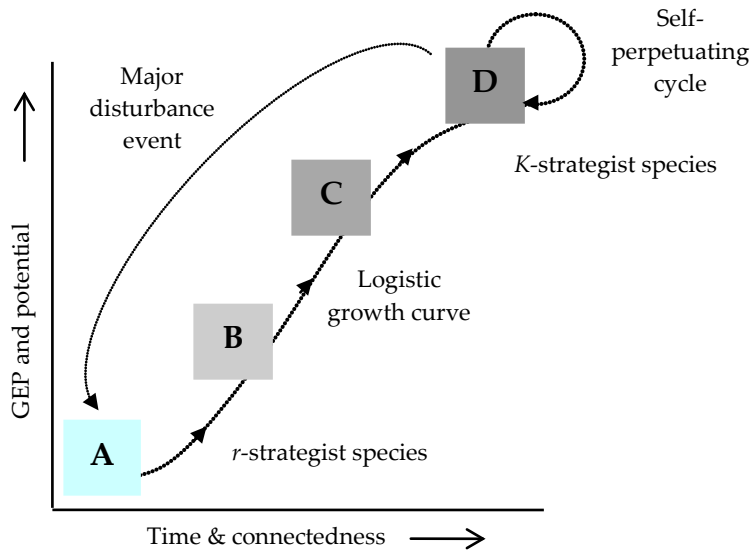

Figure 4: A stylised diagram representing a hypothetical depiction of ecosystem development according to the succession-to-climax model. Boxes **A**, **B** and **C** indicate successional phases, which develop in a linear manner towards box **D**, the climax phase.

In spite of the theoretical eloquence and clarity of the succession-to-climax model in portraying the process-functionalist approach, it has been severely criticised as being too narrow in scope, rigid and simplistic, and unable to account for and explain the scientific findings of variability in successional pathways, the constant dynamic changes in community composition and the non-equilibrium states observed with ecosystems (Hunter *et al.*, 1988). After all, the development pathway or trajectory to a climax phase is likely to require a much longer time horizon than the "natural" frequency of disturbance events (Kimmins, 1996). Moreover, we know that when interactions between species in an ecosystem are explicitly modelled, their behaviour is non-linear, which will not produce a linear system. Indeed, it is well known that even simple non-linear difference equations of single species models may produce bizarre, extremely non-linear dynamics (Gleick, 1987). Thus, it seems that the succession-to-climax model is flawed, not only because a complete ecological recovery of an ecosystem to a climax phase that maintains itself again and again can never be realised (Jorgensen, 1997), but because the non-linearity between species is almost certainly not going to lead to a linear system of the whole.

Consequently, the likelihood of ecosystems developing in a linear manner, whereby the system reaches a deterministic and predictable single steady state with homeostatic properties is highly questionable (Holling, 1986; Botkin, 1990; Kay, 1993; Schneider & Kay, 1994; Pahl-Wostl, 1995; Kay *et al.*, 1997; Kay & Regier, 2000).

It seems that ecosystems are best-described as "soft" systems. As such, assumptions whereby stability is presumed and change is explained away should in fact be altered whereby a model of ecosystem behaviour and development should implicitly assume change and explain stability (van der Leeuw, 2000). In other words, life is not in a stable equilibrium, but is dynamic and constantly changing, and as such, change cannot be thought of as an "error term", or anomaly, but rather as law. It seems the definition of ecosystem health and the subsequent management of ecosystems must acknowledge that ecosystems are dynamic and ever-changing entities. Therefore, a model of ecosystem development should incorporate the system's need to meet the challenge of changing factors, and not attempt to model the eternal "struggle" to return to exactly the same ecological community. For this reason, determining the development and behaviour of ecosystems appropriately requires that models are based on non-linear dynamics (Budiansky, 1995; Kay & Regier, 2000).

3.2 Non-linear dynamics, multiple equilibria and resilience

The non-linear nature of ecosystem development implies that multiple equilibria exist, rather than a single equilibrium, as determined in the linear succession-to-climax model (see *Figure 5*). Accordingly, given that non-linear models have multiple equilibria, ecosystem states are only locally stable, and not globally stable as assumed in a linear model. Importantly, this phenomenon of multiple equilibria is not just a mathematical artefact, as the presence of multiple ecosystem states and transitions among these states has been observed empirically in a range of ecosystems. These include transitions from grass-dominated to woody-dominated semi-arid ecosystems in Zimbabwe (Walker *et al.*, 1981) and Australia (Walker *et al.*, 1997).
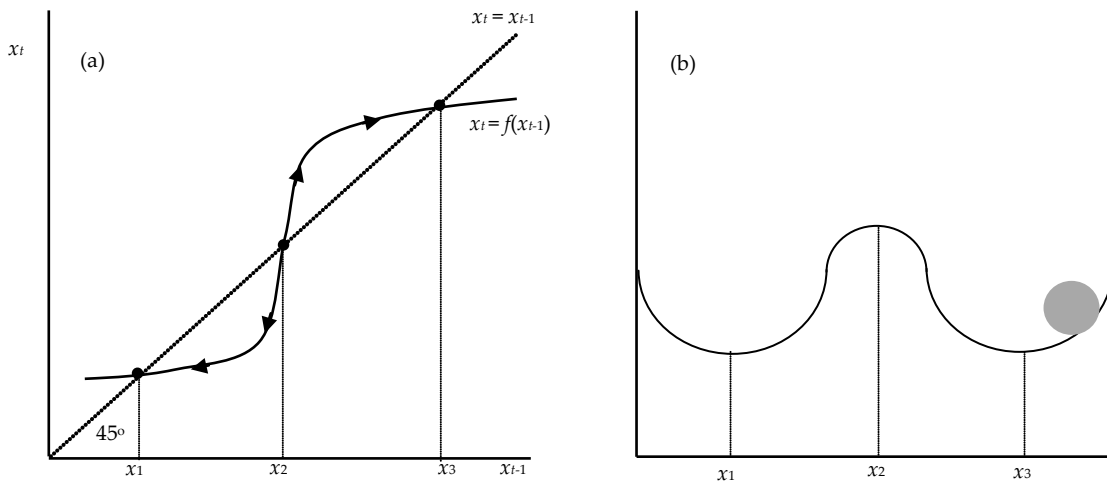

Figure 5: (a) Graphical representation in discrete time of a dynamic non-linear model. The system has multiple equilbria demarked as x_1 and x_3, while x_2 is an unstable state. (b) Ball-and-cup diagram of the same non-linear model. The cup represents a particular state of the system and the ball represents the current position of the system within the state.

In order to model the a non-linear dynamic system maintaining a particular system state despite perturbations, the concept of resilience was introduced formally by Holling (1973), who described the resilience of an ecosystem appropriately as "ecological resilience". Thus, ecological resilience is diametrically related to notions of local stability and elasticity of the system state (Ludwig *et al.*, 1997). However, while ecological resilience is related to stability, Holling (1973; 1986) warns not to mistake ecological resilience for stability *per se*, as stability is relevant when an ecosystem's state is close to its equilibrium, while ecological resilience is most relevant when investigating non-linear conditions far from its steady state. Thus, ecological

resilience is effectively about the pressure-stress-response capability of the system state, that is, the magnitude of disturbance a particular state can absorb without transitioning to an alternative system state (Holling, 1973; 1986; Holling *et al.*, 1995).

While the definition of ecological resilience seems fairly clear, an alternative definition of resilience has also been proposed (Pimm, 1984). This form of resilience, described here as "economic resilience" propounds that resilience can be measured by the speed of return of a system to its steady state following a disturbance event (Pimm, 1984; O'Neill *et al.*, 1986; Tilman & Downing, 1994; Ives, 1995; Neubert & Caswell, 1997). Thus, while ecological resilience defines resilience as the maintenance of existence of functioning, economic resilience defines resilience as the maintenance of efficiency of functioning (Costanza, 1992). *Figure 6* depicts the concepts of ecological and economic resilience.
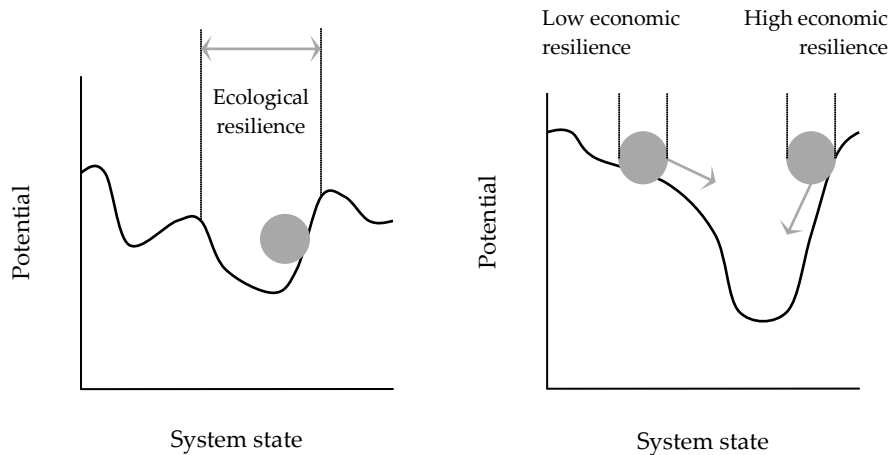

Figure 6: (a) Stability profile illustrating ecological resilience. Importantly, the width of the system state "cup" dictates the magnitude of ecological resilience. (b) Stability profile illustrating economic resilience. Depressions in the landscape with flatter slopes have less economic resilience than areas that have steeper slopes (*Source*: Gunderson *et al.*, 2002).

Economic resilience implicitly assumes that the system will always attempt to reach a steady state and the ecosystem response will be approximately proportional to the perturbation. This is of course unlikely to be so, given the non-linear nature of ecosystems. Thus, the use of economic resilience can only be justified in states where ecosystem behaviour is functioning near its steady state, and not far-from-equilibrium states. As such, economic resilience may be useful for investigating systems that operate in a linear fashion, or at least where a linear approximation is sufficiently valid (Pimm, 1991). Indeed, it could be argued that even where multiple equilibria are shown to exist, a linear approximation might be acceptable by way of argument that expectations and norms make the other equilibria unlikely. Furthermore, given that there is unlikely to be a general solution for non-linear relationships, many analysts have been content to investigate only the local stability of a particular state within a non-linear system. However, in ecosystem development, the critical feature is stability far from any equilibrium because it is in these areas that a system may make a transition into an alternative system state. Hence, only ecological resilience can be considered satisfactory.

3.3 The diversity-stability model

An implication of multiple equilibria in ecosystem state space is that no longer can we expect ecosystems to maintain global stability. Thus, it might be considered prudent to keep ecosystems in a stable state and to avoid "flips" into less stable ecosystem states. Hence, one can argue that the health of an ecosystem is reflected by its stability (or ecological resilience) (Batabyal, 1998; Ferriera & Towns, 2001). Needless to say, research for half century has focused on the long held belief that connectance (number of connections) of an ecosystem begets the stability of that ecosystem. That is, it has long been presumed by ecologists that the

greater the species diversity (a proxy measure for connectedness) present in a system, the more stable the system network is likely to be and the more likely the various ecosystem functions will be maintained (Folke *et al.*, 1996). This diversity-stability hypothesis is the most legitimate ecological argument for preserving diversity within ecosystems. Importantly, in focusing on diversity, this hypothesis is grounded in the "population-community" approach to ecology, which focuses on organisms and species, and thus views ecosystems as "networks of interacting populations" (O'Neill *et al.*, 1986; Allen & Hoekstra, 1992). However, the diversity-stability hypothesis, has been one of the most controversial and ongoing debates in ecology, whereby the relationship has been validated and rejected on numerous occasions.

Pioneering research investigating the diversity-stability hypothesis by MacArthur (1955) seemed to affirm that ecological communities that were highly connected are more stable than simpler ones. MacArthur (1955) suggested that there was a direct correlation between the logarithm of the number of food links in a food web and the degree of stability. Elton (1958) added further weight to the argument when it was pointed out that the apparent extreme stability of tropical rain forests may well be because these systems are the archetypal diverse and connected ecosystem. The observed positive relationship between connectance and stability, however, initially came under attack on mathematical grounds from May (1973) and later by Goodman (1975). May (1973), in his study of randomly assembled model food webs, found exactly the opposite: "too rich a web connectance… leads to instability". This was reasoned, because as the number of species increases, the probability increases that one of them will be associated with a real positive eigen value, which will hence act towards an unstable mode of oscillation within the system. May (1973) reconciles his arguments with the undisputed data of more oscillatory behaviour in simpler boreal ecosystems than in more species diverse tropical ones by arguing that the causation is reversed. Following the mathematical account for connectance leading to instability, came empirical evidence of the phenomena by Weiderholm (1980), who observed that increased phosphorus loading gives decreased diversity, but very stable systems.

While, it might seem that diversity and connectance decrease stability, Pimm (1991) found in computer simulations, ecosystems with few species were easy to invade and destabilise. Indeed, ecological communities of up to twelve species were easily entered by intruding and destabilising species. This conclusion was soon supported with empirical evidence by Baskin (1994; *p*.203), who concluded from his findings that the "biggest gains in stability, for example come with the first ten species in a system; beyond ten, additional species did not seem to add much stability, perhaps because the essential functional niches had already been filled". Interestingly, Baskin (1994) further noted that similar conclusions can be made for productivity, in that "more diverse systems are more productive – at least up to a point". Despite, these conclusions of the role of connectivity and diversity recently, Loreau (1999) posited that diversity is able to buffer temporal variance, and Rozdilsky and Stone (2001), in strictly competitive systems found increased diversity can lead to increased stability. Thus, again one is left with much ambiguity about what determines a stable system.

It seems from this ecological research that one is left to conclude that ecosystem stability is a very complex concept and it is likely that there is no simple relationship in which to model it. However, MacArthur (1972) suggests that stable systems have in fact intermediate levels of connectance. This credible proposition (as we will confirm later) was also taken up by O'Neill *et al.* (1986) who concluded that, because a system can become unstable either by being over or underconnected, the addition of a new component can have an effect opposite to what might be intuitively expected. Thus, an increase in diversity can stabilise the system, either by adding connected parts to an underconnected system, or removing connected parts to an overconnected system. Thus, the stability of ecosystems in its widest ecological sense might be best considered a multidimensional relationship. This diversity-stability relationship may be formulated as follows: if the system can offer a better survival (*i.e.* increasing stability in relation to the changing forces functions by decreasing the diversity), the system will not hesitate to react accordingly. Thus, the more diverse an ecosystem, does not give the best answer to stability and survival (Olmsted, 1988). Diversity is a two-edged

sword, and despite its ease of measurement, diversity indices such as the Shannon-Weaver index should not be considered a prime *de facto* measure for ecosystem health.

But, despite MacArthur's and O'Neill's conclusions and the difficulties of diversity and stability as measures of ecosystem health there still remains the significant difficulty that much of the empirical evidence examining the diversity-stability hypothesis investigated population networks close to equilibrium and not states far-from-equilbrium. Indeed, the role of diversity over a much broader range of variations and the relationship between diversity and ecological resilience has only recently been addressed, and while likely to provide a similar relationship, this is yet to be sufficiently proven (Levin, 1995; Perrings *et al*., 1995; Peterson *et al*., 1998; Gunderson *et al*., 2002). However, when attempting to unravel an ecological relationship for resilience, some observers consider Walker's (1992) driver species hypothesis credible. This hypothesis proposes that species can be divided into two groups, "driver" species and "passenger" species. Driver species are effectively "keystone species" (Paine, 1969) in that they control the resilience of the system, while the passenger species do little to maintain the system in a particular state. Thus, the basic premise of driver species hypothesis is that some species have unusual qualities in that they have much stronger non-linear interactions with the overall behaviour of the ecosystem network compared to the weaker interactions found with other species. However, despite the ease of measurement for ecosystem health (*i.e.* health simply measured by the abundance of driver or keystone species in the system state), there is little anecdotal evidence in the scientific literature to support the such a hypothesis (Mills *et al*., 1993; Bond, 2001; Wassenaar & Ferreira, 2002).

Most empirical evidence seems to suggest that patterns present in ecosystems are for the most part entirely independent of the species the ecosystem contains (Naemm *et al*., 1994; Holling *et al*., 1995; Lockwood & Pimm, 2001). In fact, ecosystem functioning can normally be preserved even as the component species normally considered responsible for that particular function are lost, as other species readily fill the vacated niche (Tracy & Brussard, 1994). Similarly, studies of various ecosystems have also shown that the population dynamics of individual species are more sensitive to stress and perturbations within ecosystems than are ecosystem processes (Schindler, 1990; Vitousek, 1990). These findings then beg the question of how important any species are in an ecosystem. In theory, we could experimentally delete species one at a time, measure the ecosystem impacts on resilience and function, and generate a frequency distribution of species importance, relative to abundance. Despite this seemingly insurmountable challenge, some ecologists have tried to do just that, but the research remains inadequate as only a fraction of the species in an ecological community have been deleted (Berlow *et al*., 1999). An additional problem is that the importance of a species might change in different places or at different times (Power *et al*., 1996). So, a species that may be highly valuable ecologically in one place and at one time may or may not be important ecologically in another place or at another time.

It seems that we remain uncertain as what is the underlying relationship to deciphering ecological resilience. Despite this difficulty, Costanza (1992; *p.*7) proposed that, a system state should be considered healthy "if it is stable and therefore sustainable; that is, if it is able to maintain its metabolic vigour, its internal organisation, structure and autonomy and is resilient to perturbations and stresses over a time and space frame relevant to the system". In effect, Costanza suggests that to model ecosystem health we must integrate measures of function, structure and stability, which he did by formulating a straightforward though *ad hoc* multiplicative index of ecosystem health by incorporating potential, connectedness and ecological resilience variables together (see *Equation 2*).

$$\max EHI \equiv C \times P \times R$$

Equation 2: An index of ecosystem health (*Source*: Adapted from Costanza, 1992; Mageau *et al*., 1995).

Where EHI is the ecosystem health index
C is the connectedness of the ecosystem
P is the potential of the ecosystem
R is the ecological resilience of the ecosystem

Regardless, of Costanza's efforts to develop a suitable ecosystem health index, it is apparent that the various models representing both the "process-functionalist" and "population-community" approaches are insufficient and have considerable reductionistic tendencies that fail to capture the dynamics of non-linear systems and explore under the mist of ecosystem "complexity". However, recently a general theory of non-linear systems first conceptualised by von Bertalanffy (1968) has emerged from outside the scientific field of ecology, which seems to have finally unveiled the development and behaviour of ecosystems into just a few fundamental principles.

4.0 Self-organisation, emergence and thermodynamics

The new theory of development and behaviour of dynamic non-linear systems is at its core based on the principle of self-organisation. That is, dynamic non-linear systems, such as ecosystems tend to lead to a process of lower to higher levels of organisation, while being kept within limits (Schuster & Sigmund, 1980). Thus, self-organisation is a process that leads to the emergence of higher formed network structures (*Figure 8*). The nexus of this self-organising process is an "attractor", whereby attractor comes from the state space description of the behaviour of the ecosystem. Thus, a state within a system behaves as if it were "attracted" toward a domain. The dynamics of a self-organising system are largely a function of internal causality and as such the system is dominated by non-Newtonian positive and negative feedback loops. These feedback loops allow the system to maintain itself about an attractor despite changes in the external environment, because the feedback loops of the system tend to maintain the system's present state. Therefore, the environment may change substantially, without the system exhibiting major change. It is this capacity to organise and maintain itself about an attractor that is the fundamental hallmark of a self-organising system. As such, a self-organising system implies a goal-like function (similar in character to the climax phase in the succession-to-climax model), whereby internal causal mechanisms direct the ecosystem towards the state attractor.
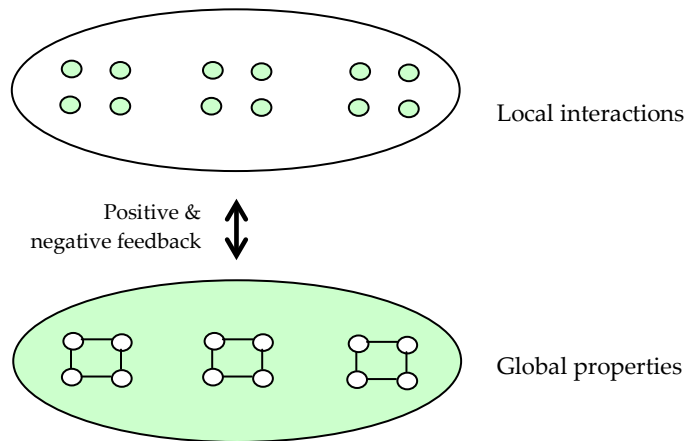

Local interactions

Positive &
negative feedback

Global properties

Figure 8: The diagram illustrates the process of self-organisation leading to emergence, where emergence is developed from an initial group of localised interacting agents, which leads to the formation of a higher network structure with global properties (*Source*: Schuster & Sigmund, 1980).

At this point, it is important to acknowledge that some academics may feel somewhat uncomfortable about the principles of self-organisation and emergence, because such notions may be likened to "vitalism", the now defunct idea that a Platonic-like "life force" is what inhabits life and nature, and which directs an ecosystem along some trajectory. However, it is important to re-enforce that self-organisation is process developed internally within the system, and is not the product of an "external engine" as vitalists would contend. Emergent properties arise from local interactions among system components, and in turn they influence the local interactions. All evolving ecosystems possess emergent properties and appear to behave like much-like super-organisms (Kay & Regier, 2000). But this super-organic behaviour is the result of a continuing two-way feedback between local interactions and global properties. It is not the outcome of some "mystical" global

property determining the local interactions of system components, nor is it solely the cumulative result of local interactions in the system that can be explained by reductionistic analysis. It truly is the system as a whole being an integrated, dynamic structure involving two-way interaction between all levels. Importantly, it is these notions of self-organisation and emergence that seem to explain ecological and biological form unexplained by the processes of natural selection alone. Accordingly, Levin (1999) postulates that the combination of self-organisation and natural selection seen to be the appropriate perspective to view evolution.

At the core of ecosystems as self-organising systems is the thermodynamic conceptualisation that such systems are dissipative systems (Prigogine, 1980). A dissipative system is one which is subject to the second law of thermodynamics. That is, according to the second law, energy with a high amount of information and organisation always dissipates in a materially closed system. Thus, while the quantity of energy is always conserved as implied in the first law of thermodynamics, the quality as implied in the second law does not, which means that all energy transformations will involve energy of higher quality being degraded to energy of lower quality.

It initially seems that the second law of thermodynamics would lead us to rather pessimistic conclusions of decay and degradation. However, we also know that decay towards thermodynamic equilibrium is it seems in stark contrast to the development of ecosystems, which seem to exhibit the propensity towards ever more higher structures with emergent global properties. But, one is left wondering how do ecosystems have this ability to build up and maintain increasingly complex structures, when the components of a system have an inherent predisposition towards disorder, decay and degradation? It would appear there is a paradox of life; that is, the emergence of complex structures in the face of the second law of thermodynamics.

This contradiction was resolved by Schrodinger (1944), who fittingly pointed out that whereas the second law describes isolated or closed systems; all ecosystems have to be described as open systems, which exchange energy with surrounding systems and their environment. Thus, an ecosystem is not strictly a system, but rather a system of systems. That is, there is a hierarchical nature of systems, whereby each system is nested within a system and is made up of systems (Allen & Starr, 1982). Accordingly, because an ecosystem is an open system it can maintain a non-equilibrium state and avoid thermodynamic equilibrium by importing high quality energy from other surrounding systems and its external environment, whilst exporting low quality energy. This exchange of entropy (*i.e.* a measure of energy disorder, whereby high quality energy has low entropy) by the ecosystem allows the ecosystem's total entropy to decrease, while inevitably increasing the entropy in the surrounding system's environment. To that end, Schrodinger (1944; *p*.75) surmised these findings by expressing that "life feeds on low entropy".

In view of Schrodinger's findings, it can be understood that self-organising dissipative processes emerge within open systems whenever a sufficient throughflow of high quality energy is available to support them. The details of these dissipative processes depend on the materials available to operate them, the energy and information present to catalyse the processes, and the surrounding environment. The interplay of these factors defines the context for the set of processes which may emerge (Jorgensen, 1997; Kay & Regier, 2000). Once a dissipative process emerges the open system has a high propensity to move away from thermodynamic equilibrium, and when the system does move it will reach a critical distance from equilibrium, whereby the open system responds with the spontaneous emergence of new organised behaviour that uses the throughflow of high quality energy to manifest and organise itself as a complex ecological structure. These structures provide a new context, nested within which new processes can emerge, which in turn beget new structures. And with more high quality energy obtained by the system, an ecosystem ultimately emerges. Thus, an ecosystem according to this new perspective of dynamic non-linear systems can be described as a nested constellation of self-organising dissipative processes and structures organised about

a particular set of sources of high quality energy, materials and information, embedded in the environment (Kay & Regier, 2000).

Once ecosystem structure is established, ecosystem growth is provided by high quality energy being stored in the formation of ordered structures of biomass. In thermodynamic terminology, this growth builds up further organisation and structure, and continues to allow the system to move against the gradient imparted by the second law of thermodynamics. Importantly, it seems that if the system is offered more than one pathway to move away from equilibrium, the organisation yielding the most growth or stored high quality energy will be selected (Jorgensen, 1997). However, for ecosystems and its species to continue to grow they must adapt and specialise to their surrounding environment, which is achieved by storing more information into the system. This process of diversification, adaptation and specialisation in turn increases the emergent structure, while allowing the system to be more efficient at utilising the system's resources so as to be more effective at building more structure and more "fitted" to the prevailing environmental conditions, which thus further enhances the dissipating capability of the ecosystem. (This transformation towards greater diversification and specialisation in nature can be explained in much the same way as Adam Smith's (1776) famous pin-making example, whereby tasks are sub-divided so as to increase returns to production). Importantly, this development towards increasing structure, through diversification, adaptation and specialisation almost seems to be a natural law of self-organising systems, akin to the second law of thermodynamics, though unlike the second law, this law acts in the reverse direction to energy degradation (Jorgensen, 1997; Kay & Regier, 2000; Puu, 2000).

A mature ecosystem is then with the increase in adaptation and specialisation of its species to its environment, a very complex highly interconnected structure. Moreover, it contains a very high concentration of biomass and contains much information in a wide variety of species. However, at this mature stage, most of the high quality energy captured by the ecosystem goes into maintaining its structure and only a very small amount into further growth. The growth of biomass finally stops when one of the essential building blocks for growth becomes scarce (Jorgensen, 1997). It seems that "all aspects of life are ultimately governed by the scarcity of resources" (Hirschleifer, 1982; *p*. 52). (Interestingly, one might then speculate that the development of human cognition has evolved so as to utilise resources more efficiently). Thus, in conclusion ecosystem development proceeds in a way that: one, captures resources (high quality energy and material); two, makes more effective use of resources; three, builds more structure; and four, enhances survivability (Schneider & Kay, 1994; Kay & Regier, 2000).

4.1 The emergy, exergy and ascendancy models

In an effort to gain some consensus on an operational measure and characterisation of ecosystem health, while acknowledging ecosystems as complex self-organising systems there it would seem is a need to make an assumption of how we perceive an ecosystem conducts itself as if it has an internal "objective function". One thermodynamic objective function proposed is that ecosystems organise themselves to maximise the degradation of the available work in incoming energy (Kay, 1991). This goal of ecosystems has been captured by the notion of emergy (Odum, 1996), which expresses the amount of energy it costs to build an ecosystem and is measured utilising energy flows, much like Hannon's (1992; 1999) novel GEP measure. An alternative thermodynamic objective function is that ecosystems attempt to maximise their storage of energy. This goal function can be measured by the notion of exergy, which is a measure of the amount of high quality energy (or information) stored in an ecosystem structure. The biological and ecological meaning of this objective can be related to Darwin's "survival of the fittest", that is, survival means growth, which is equal to increased high quality energy of the system relative to the environment (Jorgensen, 1997). The organisation that is able to produce the highest exergy under prevailing conditions will be selected.

Clearly, then the difference between the emergy and exergy models, is that while the former focuses on how much energy it costs to create the ecosystem structure, the latter considers to account for the ability of the ecosystem to do work. Importantly, the same ecosystem can have quite different emergy and exergy values, because for example, some environments will require a greater amount of energy to produce similar structures of biomass. Jorgensen (1997) suggests that exergy is a more appropriate objective function of ecosystem development because while maximum degradation of energy is a consequence of the development of ecosystems from the early to the mature state, it is not necessarily an appropriate objective function for mature ecosystems, as ecosystems cannot degrade more energy than that corresponding to the incoming solar radiation. Interestingly, exergy as a measure of ecosystem health is similar to the "entropy theory of value" conceived by thermodynamic scientist and economist Georgescu-Roegen (1971), where value is considered in part determined by the level of entropy for that object investigated. However, like Georgescu-Roegen, Jorgensen (1997) warns that while exergy capture and storage is a fundamental "objective" of ecosystem development it does not suggest that other factors should be neglected in demarcating ecosystem health. This sentiment is wise as while a measure of exergy has sound theoretical grounding, it is not only extraordinarily difficult to quantify and measure, it neglects both emergent network properties associated with dynamic non-linear systems and the need to incorporate a variable for determining the resilience properties of the system.

A network perspective of non-equilibrium systems has also been established in the traditions of the diversity-stability model, which does capture emergent network properties through modelling ecosystem behaviour by information and network theories. One promising though again somewhat impractical measure of these theories is the "ascendancy" index (Ulanowicz, 1980; 1986; Wulff *et al.*, 1989). The ascendancy index and measures like it go several steps beyond species diversity indices used in ecology, because they estimate not only how many different species there are in a system but, more importantly, how those species are organised collectively in the ecosystem. Thus, a rise in the index of ascendancy represents an increase in system size and organisation, which translates a measure of growth and development, and thus survival. However, network approaches, such as the ascendancy index are almost entirely used to investigate systems near steady state. One can conclude that to truly represent and model the principles of self-organisation and emergent global structures, both thermodynamic (energetic) and network measures of ecosystem health (*i.e.* ascendancy and exergy) must be integrated, as in fact they are complementary perspectives and measures on how an ecosystem develops and behaves (Nielsen & Ulanowicz, 2000). But, the development of this pluralistic viewpoint of so-called "thermodynamic networks" requires more research in hierarchy theory and as such is still very much in its infancy (Jorgensen, 1997; Kay *et al.*, 2001).

4.2 Order, chaos and complexity
The fundamental dynamics of non-linear systems has revealed that away from equilibrium the nature of these systems are surprisingly rich and complex, whereby non-equilibrium conditions are a source of organisation and therefore order. In general, a system near thermodynamic equilibrium, being stable, can accommodate fluctuations from the mean state. When forced to move away from equilibrium by externally applied gradients, a critical point may be reached where the fluctuations can no longer be accommodated and instead are amplified to produce a new macroscopic order, described as "complexity", the "edge of chaos". The process involves an instability being triggered by fluctuations that exceed some threshold, and the system then reorganises itself to accommodate the instability. Thus, Schneider and Kay (1995; *p*.232) write:

> Life emerges because thermodynamics mandates order from disorder whenever sufficient thermodynamic gradients and environmental conditions exist. But, if life is to continue, the same rules require that it be able to regenerate, that is create order from order. Life cannot exist without both processes, order from disorder to generate life and order from order to ensure the continuance of life.

But, as expressed by Ulanowicz (1997) there is a "window of vitality" to this macroscopic order of complexity, that is, there is a minimum and maximum level in between which self-organisation can occur and emergence is maintained. Too much or too little of each of these opposing forces (*i.e.* internal and external forces) create an imbalance which will upset the efficiency of the system. Thus, the point where the disorganising forces of the environment and the organising forces of the ecosystem are balanced, an optimum operating point is established (Kay, 1984; Kay, 1991). Indeed, if a far-from-equilibrium system such as a highly inter-connected mature ecosystem becomes isolated, and severed from its energy sources, then it will decay towards thermodynamic equilibrium by irreversible processes (Ulanowicz, 1997). Thus, there is a range within which self-organisation and emergence occurs, and as such complex self-organising systems do not strive for a maximum (as would implied in measuring emergy or exergy), but rather an optimum. Therefore, in accordance with findings of diversity-stability relationship, ecosystem development should be modelled as if it has an effective lower and upper limit.

To grasp this macro-state of complexity consider a set of species as components of an ecosystem. The specific macro-state of the system will be determined by the degree of connections that bind the species of the system together. Now presume that all possible states imaginable can be arranged along an axis. Effectively, this axis (see *Figure 9*) defines state space (Kauffman, 1993), that is, state space is the set of all possible system states that can be constructed from the given set of species available. At the ends of this axis, lie the two extremes: the null set of states, or states of order, which have no or few connections; and the complete set, or states of chaos, which have a high number of connections (or a complete set of connections). The adjacency structure of state space then distinguishes all possible *n*-systems.
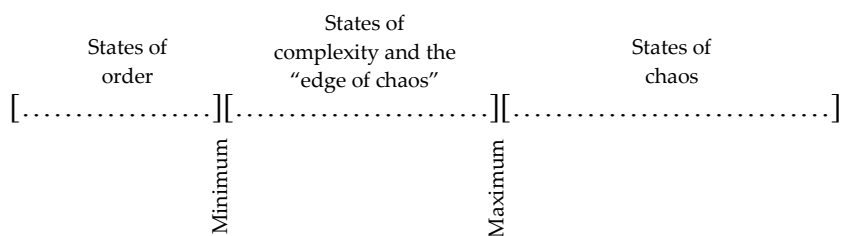

Figure 9: The axis of state space. Importantly, only states within the "states of complexity" are able to produce emergent self-organising systems.

Importantly, in states of order, coherence and stability of information is maximised, but experimental arrangements of that structure are minimised. In chaotic states the opposite applies, such that it effects a massive search of the possibilities within state space, but is unable to lock onto any that are "useful". Complexity and the states that lie within it, on the other hand, could be considered an inference of the effect that a balance between stasis (states of order) and change (states of chaos) is the ultimate principle underlying all time evolutionary and self-organising processes. Thus, while equilibrium is the expression of "balance" in a linear "economic" world, complexity is the expression of "balance" in a dynamic non-linear "real" world (Potts, 2000).

Importantly, when we consider the non-linear relationships that exist in ecosystems, we might well wonder why chaotic states are not observed more often. The obvious answer is that nature attempts to avoid chaos, so that it can continue to self-organise and evolve. Naturally then, the health of an ecosystem should be determined above all by the ability an ecosystem has in maintaining its "integrity" for continued self-organisation (Kay, 1991; Muller, 1998; Kay & Regier, 2000). Importantly, it is this definition of ecosystem health that is sometimes referred to as "ecosystem integrity" and thus, fundamentally different conceptually to ecosystem health (*e.g.* Karr, 1996). However, here we consider ecosystem integrity as simply another interpretation of the ecosystem health concept.

4.3 The adaptive cycle model

It was Holling (1986) who first developed a means to model specifically ecosystem development, with the principles of emergence, complexity, evolution and the integrity of self-organisation being intuitively incorporated. The representative model coined the adaptive cycle is structured by a sequence of four phases that occur within a system state. The first two phases of the model are similar in conception to the predictable and certain nature found with the succession-to-climax model. The first phase named the exploitation or *r*-phase begins with the ecosystem exploiting those ecosystem processes that are responsible for rapid colonisation of disturbed ecosystems during which developmental *r*-strategist species capture easily accessible resources. The second phase, described as the conservation phase occurs when the slow resource accumulation builds and stores increasingly complex structures, whereby *K*-strategist species predominate, hence this phase is also described as the *K*-phase of the cycle. Ecological capital consisting of biomass and physical structure increases during the long periods of the slow dynamic sequence from exploitation to conservation (*r* to *K*), while at the same time the system state becomes more and more tightly bound with existing species. Thus, the ecosystem's connectedness increases to a point, eventually becoming too rigid and over-connected (Holling, 1986; 2001).

At a certain point, the tightly-bound accumulation of ecological capital becomes too fragile through the ecosystem being "overconnected". The actual change from the *K*-phase to the third phase, "release" or the Ω-phase is triggered by agents of disturbance. The disturbance suddenly releases the resources accumulated and sequestered as ecological capital and the tight organisation is lost. Importantly, the Ω-phase is sometimes referred to as "creative destruction", which is a term originated by the economist Schumpeter (1954; 1964), to explain alterations in the economy between periods of renewal and periods more conducive with the predictability modelled in economies following a strictly neo-classical trajectory (*r* to *K*). Finally, the process of change resultant from the Ω-phase creates opportunity for the fourth phase, reorganisation (α-phase), where released ecological capital is mobilised to become available for the next *r*-phase.

In contrast, to the *r* to *K* stage, the Ω to α stage contains considerable uncertainty. At that stage, the previously accumulated mutations and capital can become reassorted into novel combinations, some of which nucleate new opportunity and where new species can evolve. Importantly, once the system reaches the α-phase, if the system still retains sufficient amounts of its previous components it can reorganise to remain within the same state as before. However, if the reorganisation process in the α-phase does not retain sufficient amounts of previous components it may make a transition into an alternative system state *x*. Thus, it is as if two separate "objectives" are functioning, not simultaneously but instead in sequence (Holling & Gunderson, 2002). The *r* to *K* stage maximises production and accumulation; then the Ω to α stage maximises invention and reassortment. *Figure 10* illustrates a stylised representation of the four phases of the adaptive cycle illustrated within the two dimensions, potential and connectedness.
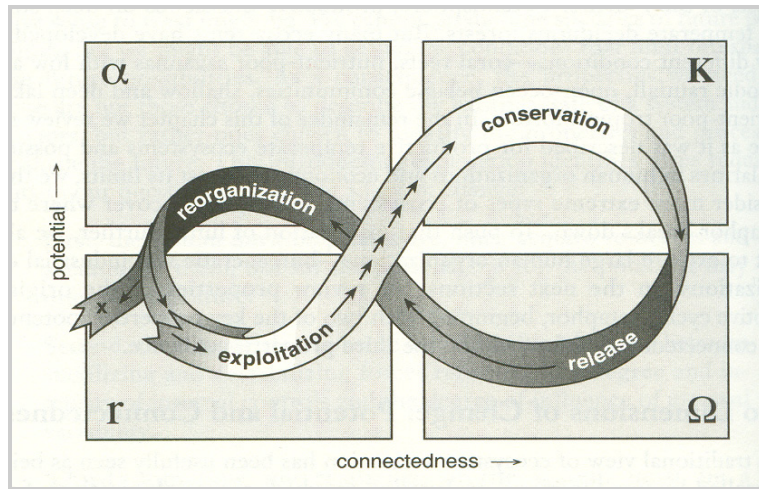
Figure 10: Stylised representation of Holling's adaptive cycle illustrating the four phases of ecosystem development (*r, K, Ω, α*) and the dynamics between these phases. The arrows indicate the speed of the development in the adaptive cycle, where short arrows indicate slow change and long arrows indicate fast change. The above adaptive cycle diagram is plotted against two ecosystem variables: 1. Y axis, is the potential of the system, which represents the ecological capital the system accumulates as well as unexpressed random genetic mutations (ecosystem innovations); 2. X axis, is the degree of connectedness (*Source*: Holling, 1986; Holling & Gunderson, 2002).

Figure 11 adds the third dimension, ecological resilience to the adaptive cycle. This orientation of the figure illustrates to us that as the phases of the adaptive cycle develop; the ecological resilience of the system expands and contracts. The conditions that occasionally foster novelty and experiment occur during periods in the back loop of the cycle, when connectedness is low and resilience is high (that is, during the *α*-phase). The low connectedness permits novel reassortments of elements that were previously tightly connected to others in isolated sets of interactions. Interestingly, there is a hypothesis, as yet unproven, that the slow variables (those with the largest inertia or *r*-to-*K* phase), rather than the fast variables, are responsible for the resilience properties of an ecosystem (Holling & Gunderson, 2002).
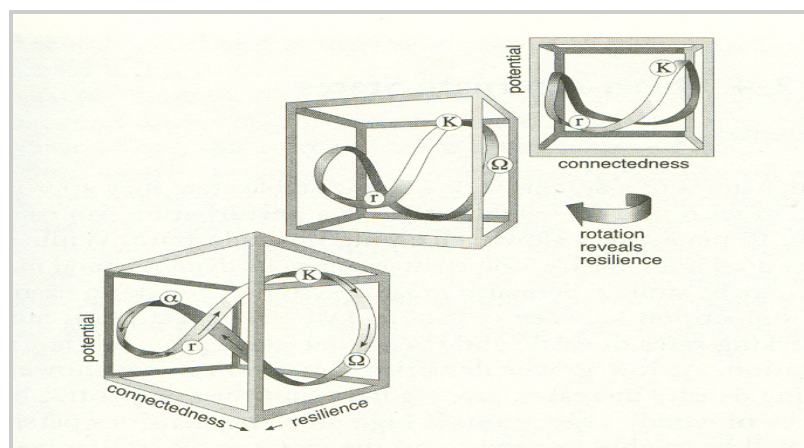

Figure 11: Ecological resilience is another dimension of the adaptive cycle and is added to the two dimensional diagram shown in *Figure 10* (*Source*: Holling & Gunderson, 2002).

An important implication of the adaptive cycle model and the integrity of self-organisation in general, is that because of nature's cyclic and self-organising behaviour one cannot identify a goal function of ecosystem development. Hence, there is no phase or specific assemblage of species within a system state that is ecologically more important than another. As such, we cannot assume that certain species (*e.g.* driver/keystone species, *K*-strategist species or mature species) within a particular system state are ecologically better. Ecosystem health demarcated by the integrity of self-organisation and modelled by the adaptive cycle model puts species in effect on an ecologically "equal" pedestal. However, this notion of species equality should not be confused with the diversity-stability model, as the adaptive cycle model does not propound that diverse systems are ecologically "better" either.

Thus, while an impoverished system state, which has its health eradicated through mismanagement, may have low connectedness, low potential and low resilience (a position within a state referred to as a "poverty trap", though ironically such a system is unlikely to remain trapped for a long period, as the low resilience means it is highly susceptible to state space transitions), a system with a high connectedness, high potential and high resilience (see Costanza's index of ecosystem health *p.*12) may be equally maladapted. Indeed, a system with high connectedness, high potential and high resilience is effectively constrained within a "frozen" or "locked" state, described by Holling and Gunderson (2002) as a "rigidity trap", which not only prevents the system state from self-organisation, but again suggests such a system within a state is intuitively ecologically better, yet we know this is not the case. In fact, such a system if it ever came in to being would be highly brittle, and would require substantial resources to keep it maintained. Indeed, forcing systems through management to persist within existing structures may seem to avoid "problems" that occur with change. However, the longer the system is "locked" into a particular community the greater its vulnerability and the more "devastating" the collapse of the system state will be perceived to be. That is, interrupting the adaptive cycle through intensive management not only interferes with the "normal" cycle of life, but also amplifies the magnitude of change within a system so much that it may shift violently into an alternative system state (Holling *et al.*, 1995).

4.4 Catastrophes, bifurcations and uncertainty

We know that despite ecosystems being able to maintain their present system states that change in a system state is inevitable. Indeed, beyond a critical threshold value, which is poised at the edge of chaos (Kauffman, 1993), the organisational capacity of a system is overwhelmed and the behaviour of the system becomes highly unstable, whereby the system state inevitably leaves its present domain of self-organisation and attraction. At this point the system may make a transition or "flip" from one attractor to another. Importantly, these shifts are not a gradual, smooth and continuous passage between system states, but rather rapid, catastrophic and step-wise (Perrings & Pearce, 1994). Indeed, the notion of self-organised criticality seems to explain why some processes lead to a minor event, while at other times the same processes lead to major catastrophes (Bak, 1996). To see this, an appealing analogy is that of a sand pile, representing if you will the development of an ecosystem. Imagine that a thin stream of sand is being run onto a round plate. Obviously, over time a sand pile steadily develops, soon reaching the edge of the plate. The initially low pile of sand soon gets higher and higher, until suddenly more sand may trigger a small sand slide, and then say a big one, that is too say the same magnitude of disturbance (another grain of sand) may lead to a response of all size ranges. Thus, the sand pile, when it can take no additional sand before sand slides will continuously occur, represents an ecosystem poised at this critical threshold.

The actual change to another basin of attraction, that is, a new state of an ecosystem, is most often modelled by either catastrophe theory or bifurcation theory. Catastrophe theory was originally developed by the mathematician Thom (1975) and explains state transitions in a way that a system trajectory along a smooth surface will at certain points have combinations of impossibility, which correspond to "folds" in the surface mapping. Thus, a system approaching one of these "folds" must make a jump, in so doing the system faces

what Thom identifies as a catastrophe. Nonetheless, despite the usefulness of catastrophe theory it has scope limitations, and consequently bifurcation theory is considered the most applicable theory for modelling state transitions. The general approach of bifurcation theory is to construct bifurcations, that is, critical points whereby the trajectory of a system is divided into new possible pathways, so as to explain the state dynamics of a system (see *Figure 12*). Generally, there will be successive bifurcations as the system moves further from equilibrium, each associated with a distinct system configuration.
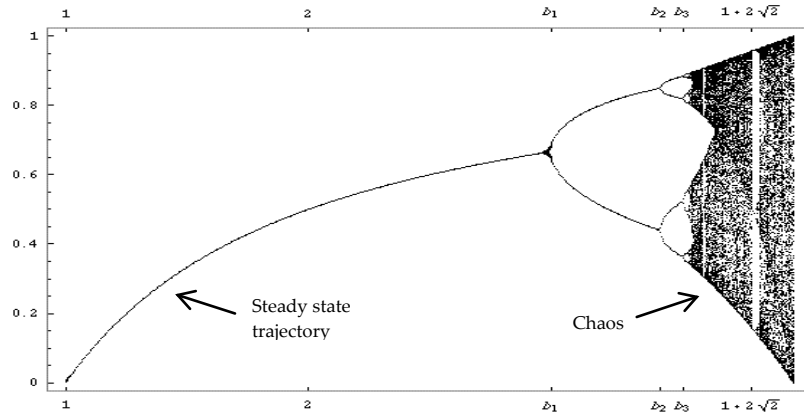

Figure 12: A typical diagram illustrating bifurcation theory.

Where b_1, b_2 and b_3 represent bifurcation points

There is however, an element of irreducible uncertainty about what new trajectory (or state) after the bifurcation will be selected prior to the actual selection occurring. This uncertainty intrinsically limits the capacity to predict categorically how a situation will unfold, say after changes in the management of an ecosystem (Costanza & Cornwell, 1992; Ludwig *et al*., 1993; Kay & Regier, 2000; Limburg *et al*., 2002). Thus, while the "time evolution" of an ecosystem is governed by somewhat deterministic and predictable laws between bifurcations (Levins, 1999), the behaviour at a bifurcation has elements of historical happenstance, which is largely unpredictable and cannot be reduced to probabilistic estimates, no matter how much information we have and how sophisticated our simulations might be. In other words, in the vicinity of a bifurcation, fluctuations with a chance-like character play a dominant role in determining the future state of the system. The reason for this chance-like character is because beyond a bifurcation a system may adopt more than one new state. Thus, changes in the system cannot be tied categorically to any specific environmental changes.

In consequence of bifurcation theory, it seems that the Laplacian aspiration to be able to quantitatively predict, with certainty, how the future will unfold, is irreconcilable with modelling of complex self-organising systems (Kay & Regier, 2000). What is certain, however, is that before the state attractor is ever reached, the state conditions, determined by the external factors and the internal ecosystem components, will have changed and a new attractor is then effective. And before this new state attractor can ever be reached, new external and internal conditions will again emerge, and so the process goes on (Jorgensen, 1997).

5.0 *The subjective social utility model*
From the work presented hitherto on models of ecosystem health the following conclusions can be delineated. First, the credibility of the naturalness model is dubious, as there are numerous philosophical and pragmatic difficulties in implementing naturalness, not least because humans and therefore human action that alter "natural" systems should in themselves be considered "natural". Secondly, the conventional approach to the scientific process that of reductionism, effectively propounds that ecosystem health can be determined by genetic diversity, however, this approach wrongly assumes that nature and ecosystems are mechanistic,

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which can be easily broken down into its component parts. Thirdly, the classical systems approach of succession-to-climax is also unconvincing as a model of ecosystem health, as it wrongly insinuates that ecosystems develop in a linear manner towards a self-perpetuating single steady state, rather than assuming the presence of multiple equilibria and non-linear dynamics. And fourthly, there is little empirical evidence that supports the diversity-stability and keystone species hypotheses, thus, ascertaining health of an ecosystem by its diversity or abundance of an exceptional species is unfounded. These conclusions demonstrate that when demarcating ecosystem health we must avoid commonly held hypotheses like: naturalness is "good", diversity is "good", extinction of species is "bad", stability and resilience is "good", and change is inherently "bad". Indeed, the most pervasive misappropriation when demarcating a concept such as ecosystem health is the insertion of subjectivity and values under the supposed objective guise of scientific research (Kapustka & Landis 1998; Lackey, 2001).

In addition, to relinquishing these habitual beliefs as to what is "good" ecologically, we must as concluded previously accept that there is no ecologically "better" assemblage of species within a system state. Furthermore, and even more critically we also must conclude that there is no ecologically "better" state within an entire ecosystem. Moreover, we also can conclude that there is no reason to believe that evolution is inherently "good" either. Indeed, the philosopher Moore (1903) like Kuhn (1962), have reasoned that there is no evidence that nature necessarily evolves toward "good". To be "better" does not necessarily mean to be more evolved; to be more evolved does not necessarily mean to be "better". Consequently, categorical statements about an ecologically "good" state or community for a given system cannot be deduced from "objective" scientific arguments. Science cannot resolve which community, state or basin of attraction is best. A value-free desire for a strictly scientific demarcation of ecosystem health, which can be applied irrespective of circumstances, cannot be satisfied at present (Regier, 1993; Kay & Regier, 2000). As such, if strictly scientific demarcations of ecosystem health cannot be satisfied, then it would seem that the most appropriate means to characterising the concept is through a set of criteria which reflect the subjective values of society. For that reason, we must treat the health of an ecosystem not as a "positive" concept, but as a "normative" concept, because ultimately society has to decide what state is considered "good" (Costanza, 1992; Sagoff, 1995). Science can only provide information, so as to inform about the management tradeoffs each state may represent.

Naturally, this logical though somewhat controversial conclusion may lead scientists to dismiss the concept of ecosystem health altogether. In truth, many scientists believe that the utilisation of value-based definitions for interpreting the phenomena of nature, while useful in general conversation may be hopelessly difficult to quantify (Ryder, 1990; Lancaster, 2000). It would seem that some members of the scientific community remain steadfast in preserving definitions of ecosystem health to those empirically "observed" in nature. While such stands plague the progress of demarcating ecosystem health, it is understandable as a scientist's learning instils an unwillingness to tarnish analysis with subjective values, as this would serve only to dilute the "objectivity" of science and potentially lead to nature being the grotesque result of human invention, imagination and fantasy (Kapustka & Landis, 1998; Lackey, 2001). Indeed, scientists may dread that given that most individuals of society prefer on the face of it industrialisation to "nature", that nature as we conceive of it may be reduced to "rubble". But, as a society when making value-laden decisions in a free market and democratic environment, we must accept the values of society whatever they maybe. Indeed, to suggest that some of the societal satisfactions that nature brings are morally superior to others only reflects certain biases (Callicott, 1992; 1998). We must as certain axioms of economic theory propound, treat all societal values and preferences concerning resource use as morally equivalent and thus, decisions made concerning resource use should be determined solely in a market environment (Randall, 1988).

Since we can and "must" actively manage nature to maintain and preserve it (Swanson, 1995), and since there is no scientific basis for demarcating ecosystem health, the possibility of managing nature for objectives that

benefit society directly come into thinking. Indeed, we can conclude that humans may not only co-exist with healthy ecosystems, but that they may alter them so as to improve upon them (Regier, 1993). Even, Leopold (1939*b*; *p*.294) an unfaltering conservationist all his life could envision these ideals when he prophetically stated; "when land does well for its owner, and the owner does well by his land; when both end up better by reason of their partnership, we have conservation. When one or the other grows poorer, we do not". Importantly, one can assume that Leopold is not suggesting that nature has actually been improved upon by industrialisation in the past, but rather that societal values and preferences made within a utility framework might actually improve upon nature as well as society in the future. One then can suppose that the economic questions of supply and demand (and costs and benefits) are what truly matters. That is, "what kind of ecological garden do we as a society want?" And, "what kind of ecological garden can we as a society get?"

In taking this economic approach we also know it is not the ecosystems as "things-in-themselves" that have "end-value" to society, rather it is a multi-attribute bundle of various ecosystem goods and services (see *Table 1*) that ecosystems supply to society that have value (or more appropriately utility) to society (Lancaster, 1971; Lockwood, 1996; Hoehn *et al.*, 2003). Thus, when demarcating ecosystem health by the aggregation of utility, it is important to grasp that this is not suggesting that managing an ecosystem is the same as managing a firm producing a multiple array of products. Indeed, the producers of a firm almost certainly have much better information of their products that they can satisfactorily produce. In contrast, we are likely to underestimate the goods and services provided by ecosystems, because of the inherent uncertainty involved in ecosystem development and behaviour (Hoehn *et al.*, 2003). Moreover, while it might be argued that if it is some ecological service the ecosystem performs that is considered important to society, then a "factory" (*i.e.* firm) might suffice that can produce the same service as supplied by the ecosystem (that is, it might be possible simply to substitute ecological capital for human-made capital) (Sagoff, 1995). But, this deduction neglects that in order for an ecosystem to function they must be treated as a whole. In essence, any weighting and aggregating of values into a social utility function must also reflect ecosystems in their entirety. As such, a social utility function is useful only in distinguishing amongst states, but not ecological communities within a state, as the integrity of the process of self-organisation and evolution within a state must be maintained.

Goods & services	Ecosystem functions	Examples
Genetic resources	Sources of unique and ever-evolving genetic information	Genes for pathogen resistance, technology for breeding
Other raw materials	That portion of gross primary production extractable as raw materials	The production of timber, fuel, and fodder
Climate regulation	Regulation of global temperature and precipitation at global or local levels	Greenhouse gas regulation, DMS production affecting cloud formation
Gas regulation	Regulation of atmospheric chemical composition	The carbon dioxide-oxygen balance and ozone levels for UVB protection
Water regulation	Regulation of hydrological flows	Provisioning of water for industrial processes or transportation
Pollination	Movement of floral gametes	Provisioning of pollinators for the reproduction of plant populations
Biological control	Trophic-dynamic regulations of populations	Predator control of prey species
Regulation of human diseases	Ecosystems can change the abundance of human pathogens	Cholera and abundance of mosquitoes can be altered
Waste treatment	Recovery of mobile nutrients and removal or breakdown of excess nutrients and compounds	Waste treatment, pollution control, detoxification
Recreation	Providing opportunities for recreational activities	Eco-tourism, hunting and other recreational activities
Heritage	Providing opportunities for non-commercial uses	Historical, aesthetic and educational values

Table 1: The bundle of value-laden goods and services (attributes) provided by ecosystems that society might value (*Source*: Constanza *et al.*, 1997; *p*.254; Curtis, 2004; *p*. 164).

In spite of the apparent ease with which a social utility function might demarcate ecosystem health, for many economists the idea of a social utility function is in itself problematical. Indeed, it is well-known in social welfare theory that it is unworkable to obtain a "true" social preference as outlined in Arrow's (1951) Impossibility Theorem. However, this obstacle can be overcome if one considers the concept of utility defined by marginal classical economists as opposed to utility defined by present-day neoclassical economists (Cooter & Rappoport, 1984). In particular, the classical economist Pareto (1897) distinguished between "utility" and "ophelimity", whereby utility was considered directly comparable across individuals in a cardinal sense and thus "objective", whilst ophelimity refers to ordinal preferences that are therefore entirely subjective and as such are not comparable amongst individuals. To illustrate the contrast between these value forms, Pareto (1896) suggested that bad-tasting medicine has utility for sick people, but not ophelimity. Thus, the difference between utility and ophelimity is the difference between being socially "useful" and individually "desired". As such, at the level of individual for socially useful things is construed as being those things conducive to human health. Accordingly, those goods and services that fall clearly into the category, can include the bundle of attributes provided by ecosystems would then constitute the components of cardinal utility and not the ordinally derived ophelimity.

A simple theoretical framework with a high level of abstraction of the social utility model is developed. The framework begins by assuming that society wants to maximise its "utility" (and therefore ecosystem health) from ecosystem management subject to certain budgetary constraints. That is, the ecosystem manager wants to maximise utility for society per dollar spent of public funds. Now suppose we were to consider n ecosystem goods and services according to those outlined in *Table 1*, which are then measured as scores x_i, for $i = 1,…n$. Accordingly, each ecosystem in its present state (or phase) would represent a Lancasterian bundle of performance scores, and thus each ecosystem state j could be represented as a vector of values a_{ij} for $i = 1,…m$. Society would then, by acquiring a collection of quantities q_j of the ecosystems, obtain a combined representation of the performance dimensions. Suppose there are in all m different potential ecosystems, which present a collection or portfolio of ecosystems from which society can derive utility from. Accordingly, given budgetary constraints it will sometimes be optimal to disinvest in one ecosystem and invest into another, "that is, it will be socially optimal to engage in conversions between assets to equilibriate returns…" (Swanson, 1994; p.805). Thus, if we put this all formally, the overall performance of the collection or portfolio of ecosystems would first be written according to *Equation 3*:

$$x_i = \sum_{j=1}^{j=m} a_{ij} q_j \;\; i = 1,…n$$

Equation 3: Performance of the collection of ecosystems.

Now suppose that utility for each system state is known and that society wishes to maximise utility according to the following social utility function in *Equation 4* subject to *Equation 3* and *Equation 5*:

$$\max \int_{t=0}^{\infty} U(x_1,…x_n,t)e^{-\beta t}\partial t = W$$

Equation 4: Social utility function.

Where W is the present value of future flows of utility

$$\sum_{j=1}^{j=n} p_j q_j = C$$

Equation 5: Budgetary constraint.

Where p_j denote the prices rendered for ecosystem management
C denotes a given fixed budget allocated by society to ecosystem management.

If we now had prices for the x_i variables we could go ahead with analysis, doing things such as evaluating different portfolios and choosing among them. The x_i variables would be included in the social utility functions and so establish demand for various ecosystems.

5.1 Change, foresight and adaptation

Marshall (1949; *p.xii*) once stated that "the Mecca of the economist lies in economic biology rather than in economic dynamics… But biological conceptions are more complex than those of mechanics". These words remain prophetic, as while we have defined a simplistic theoretical description of an optimal ecosystem management strategy through maximising social utility, we have only found the most desirable and efficient states of a portfolio of ecosystems at a point in time. However, as we are now well aware fluctuations and change within and between system states are inevitable, which in turn, will alter society's supply of ecosystem goods and services (utility). Thus, with complex self-organising systems that "change", one should not be focusing on efficiency at a point in time, but rather efficiency at future points in time. In other words, "change" cannot be modelled appropriately by being ecologically well-adapted to the conditions of the given environment and by means of optimal resource allocation, instead, "change" is best modelled as the ability the system has in adapting to possible changes in the environment (Potts, 2000). However, despite these problems with fluctuations, fluctuations within a state, which reflect "boom" (r to K) and "bust" (K to Ω) patterns in the adaptive cycle maybe intuitively smoothed. This is reasoned because informed humans through the use of foresight, anticipation and intentionality may be able to reduce the "boom" and "bust" nature of the adaptive cycle, by transmitting future scarcities into current prices (Solow, 1973).

Nonetheless, fluctuations resulting in state transitions must be considered differently, in that no matter how well informed and far-sighted an ecosystem manager might be there is no means to predicting which state configuration might eventuate after a catastrophic event. Nevertheless, as an ecosystem moves further away from the neighbourhood of its stable equilibrium, the relevant value concepts of importance shift from focusing on further utility generation to one of utility maximisation and uncertainty, because (presumably) the K-phase provides the most utility to society, but has the least ecological resilience (Ludwig *et al*., 2002). Importantly, the uncertainty of a desired system state can be determined or modelled in one sense or another, by the state's ecological resilience. That is, resilience R acts in much the same way as a probabilistic function does when determining expected utility and thus, R_j is the ecological resilience for the present system state j expressed as a probability, while $1 - R_j$ is the probability of a state transition of j.

Accordingly, it has been argued that a means to ensuring that a desirable state (or phase) is maintained is to enhance the ecological resilience of the system state, while decreasing the resilience of less desirable states (Walker *et al*., 2002). However, this approach to ecosystem health is problematic for two reasons. First, because altering resilience will affect the ability of the system to properly self-organise and may lead to unwanted "lock-in". And secondly, because we must not assume that societal values and preferences are fixed and given, but also can change (at quite possibly a faster rate of change) as well. In other words, a desirable state with "enhanced" resilience at time t may become decidedly undesirable soon after at time t+1. Yet, not only is it difficult to destabilise the once desired system into an alternative more desirable state, but there can be no assurances that this now desired state will then eventuate. As a result, the management of system states by altering their resilience for state change or adhering to a risk-averse "safe minimum standard" is not seen as effective.

It seems given the limitations imposed by complex self-organising systems, management strategies for ecosystems must focus on maintaining the capacity to adapt to ever-changing ecological conditions (Reid, 1994). That is, we must truly accept chance and not relegate such phenomena through some grossly simplified "error term" (Mirowski, 1989). As such, we must accept the fact that the more we adapt ourselves to a given environment, the less we are suited to changed circumstances. And the more we prepare for all possible

contingences, the less we have devoted ourselves to the present environment. Accordingly, how should we as a society decide the balance between the known present and the set of all possible and imaginary states and futures, when the future states remain uncertain? This is one of the fundamental problems that make up the foundations of economics (Potts, 2000). One strategy that seems to fit this problem is adaptive management, which is based in the ideologies of co-evolutionary development (Holling, 1978; Gunderson *et al.*, 1995). In adaptive management, differences between how the future actually unfolds and how it was envisioned are seen as opportunities for learning. Thus, the aim of adaptive management is to address the inherent uncertainty found in ecosystem development and behaviour by achieving management objectives while simultaneously gaining knowledge and continually updating societal values (Walters, 1986). In other words, the only appropriate means to proceed in an environment that is so uncertain is not to predict what the future holds, or to resist future system changes, but to experiment and to react to changes as they become apparent (Lempert *et al.*, 1996). This is the same way nature itself achieves "balance", that is, through maintaining and self-organising itself in states of complexity.

6.0 Conclusion

It might have been thought that ecosystem health, an ecological concept could be demarcated by ecological means, but this seems not to be so. By demystifying ecosystem health it was concluded that the concept is fundamentally "normative" and can be determined by economic means alone. But, while the concept may be best modelled by an economic paradigm through a social utility function, it must be done so in a way that also captures the intrinsic ever-changing dynamics that are quintessential to a complex, self-organising system.

7.0 References

Allen, T. F. H. and Starr, T. B., 1982. *Hierarchy: Perspectives for Ecological Complexity*. University of Chicago Press: Chicago.

Allen, T. F. H. and Hoekstra, T. W., 1992. *Toward a Unified Ecology*. Columbia University Press: New York.

Anderson, J. E., 1991. A conceptual framework for evaluating and quantifying naturalness. *Conservation Biology*, 5: 347-352.

Angermeier, P. L. and Karr, J. R., 1994. Biological integrity versus biological diversity as policy directives: Protecting biotic resources. *Bioscience*, 44: 690-697.

Aronson, J., Floret, C., Le Floch, E., Ovalle, C. and Pontanier, R., 1993. Restoration and rehabilitation of degraded ecosystems in arid and semi-arid lands. *Restoration Ecology*, 1: 8-17.

Arrow, K. J., 1951. *Social Choice and Individual Values*. Wiley: New York.

Attfield, R., 1994. Rehabilitating nature and making nature habitable. *In* Attfield, R. and Belsey, A. (*eds.*). *Philosophy and the Natural Environment*. Cambridge University Press: New York.

Bak, P., 1996. *How Nature Works: The Science of Self-Organized Criticality*. Springer-Verlag: New York.

Baskin, Y., 1994. Ecologists dare to ask: How much does diversity matter? *Science*, 264: 202-203.

Batabyl, A. A., 1998. On some aspects of ecological resilience and the conservation of species. *Journal of Environmental Management*, 52: 373-378.

Berlow, E. L., Navarrete, S. A., Briggs, C. J., Power, M. E. and Menge, B. A., 1999. Quantifying variation in the strengths of species interactions. *Ecology*, 80: 2206-2224.

Bohm, D., Hilay, B. and Kaloyerou, P. N., 1997. An ontological basis for quantum theory. *Physics Report*, 144: 323-329.

Bond, W., 2001. Keystone species – Hunting the snark? *Science*, 292: 63-64.

Botkin, D., 1990. Discordant Harmonies: *A New Ecology for the Twenty-first Century*. Oxford University Press: New York.

Budiansky, S., 1995. *Nature's Keepers*. The Free Press: New York.

Burley, F. W., 1988. Monitoring biological diversity for setting priorities in conservation. *In* Wilson, E. O. (*ed.*). *Biodiversity*. National Academy of Sciences.

Callicott, J. B., 1995. A review of some problems with the concept of ecosystem health. *Ecosystem Health*, 1: 101-112.

Callicott, J. B., 1996. Do deconstructive ecology and socio-biology undermine Leopold's Land Ethic? *Environmental Ethics*, 18: 353-372.

Callicott, J. B., 1998. The wilderness idea revisited. *In* Callicott, J. B. and Nelson, M. P. (*eds.*). *The Great New Wilderness Debate*. University of Georgia Press: Athens.

Callicott, J. B., 1999. Current normative concepts in conservation. *Conservation Biology*, 13: 22-25.

Calow, P., 1992. Can ecosystems be healthy? Critical consideration of concepts. *Journal of Aquatic Ecosystem Health*, 1: 1-5.

Caraher, D. and Knapp, W. H., 1995. Assessing ecosystem health in the Blue Mountains. *In* U. S. Forest Service (*eds.*). *Silviculture: From the Cradle of Forestry to Ecosystem Management*. General Technical Report SE-88: North Carolina.

Carpenter, S. R., Brock, W. A. and Hanson, P., 1999. Ecological and social dynamics in simple models of ecosystem management. *Conservation Ecology*, 3: 4.

Caughley, G., 1989. New Zealand plant-herbivore systems: Past and present. *New Zealand Journal of Ecology*, 12: 3-10.

Clements, F. E., 1916. *Plant Succession*: *An Analysis of the Development of Vegetation*. Carnegie Institute: Washington.

Commoner, B., 1971. *The Closing Circle*. Knopf: New York.

Cooter, R. and Rappoport, P., 1984. Were the ordinalists wrong about welfare economics? *Journal of Economic Literature*, 22: 507-530.

Costanza, R., 1992. Toward an operational definition of ecosystem health. *In* Constanza, R Norton, B. and Haskell, B. (*eds*). *Ecosystem Health*: *New Goals for Environmental Management.* Island Press: Washington.

Costanza, R. and Cornwell, L., 1992. The 4P approach to dealing with scientific uncertainty. *Environment*, 34: 12-20.

Costanza, R., 1993. Ecological economic systems analysis: Order and chaos. *In* Barbier, E. B. (*ed*). *Economics and Ecology*: *New Frontiers and Sustainable Development*. Chapman and Hall: London.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R., Paruelo, J., Raskin, R. G., Sutton, P. and van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. *Nature*, 387: 253-260.

Curtis, I. A., 2004. Valuing ecosystem goods and services: A new approach using a surrogate market and the combination of a multiple criteria analysis and a Delphi panel to assign weights to the attributes. *Ecological Economics*, 50: 163-194.

Darwin, C. E., 1857. *On the Origin of Species by Means of Natural Selection*; *or the Preservation of Favoured Races in the Struggle of Life*. John Murray: London.

Darwin, C. E., 1871. *The Descent of Man and Selection in Relation to Sex*. John Murray: London.

Davies, P., 1987. *The Cosmic Blueprint*. Unwin: London.

DeAngelis, D. L. and Waterhouse, J. C., 1987. Equilibrium and nonequilibrium concepts in ecological models. *Ecological Monographs*, 57: 1-21.

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

Depew, J. D. and Weber, B. H., 1996. *Darwinism Evolving*. MIT Press: Cambridge.

Ehrlich, P. R. and Ehrlich, A. E., 1992. The value of biodiversity. *Ambio*, 21: 219-226.

Elton, C., 1958. *The Ecology of Invasions by Animals and Plants*: *Problems in Population Biology*. Clarendon Press: Oxford.

Fairbrother, A., 1998. Establishing the health of ecosystems. *In* Cech, J. J., Wilson, B. W. and Crosby, D. G. (*eds*). *Multiple Stresses in Ecosystems*. Lewis Publishers: Boca Raton.

Ferraro, P. J. and Simpson, R. D., 2002. The cost-effectiveness of conservation payments. *Land Economics*, 78: 339-353.

Ferriera, S. M. and Towns, D. R., 2001. *An ecosystem approach to maximize conservation of indigenous biological assets of New Zealand*. Unpublished report, Department of Conservation: Auckland.

Folke, C., Holling, C. S. and Perrings, C., 1996. Biological diversity, ecosystems, and the human scale. *Ecological Applications*, 6: 1018-1024.

Folke, C., 1999. Ecological principles and environmental economic analysis. *In* Van den Bergh, J. C. J. M. (*ed*). *Handbook of Environmental and Resource Economics*. Edward Elgar: Cheltenham.

Gailbraith, J. K., 1987. *A History of Economics*. Penguin: London.

Gaudet, C. L., Wong, M. P., Brady, A. and Kent, R., 1997. How are we managing? The transition from environmental quality to ecosystem health. *Ecosystem Health*, 3: 3-10.

Georgescu-Roegen, N., 1971. *The Entropy Law and the Economic Process*. Harvard University Press: Cambridge.

Gleick, J., 1987. *Chaos*. *Making a New Science*. Viking: New York.

Goodman, D., 1975. The theory of diversity-stability relationships in ecology. *Quarterly Review of Biology*, 50: 237-266.

Gunderson, L. H., Holling, C. S. and Light, S. S., 1995. *Barriers and Bridges to the Renewal of Ecosystems and Institutions*. Columbia University Press: New York.

Gunderson, L. H., Holling, C. S., Pritchard, L. and Peterson, G. D., 2002. Resilience of large-scale resource systems. *In* Gunderson, L. H. and Pritchard, L. (*eds*). *Resilience and the behaviour of large-scale systems*. Island Press: Washington.

Hannon, B., 1992. Measures of economic and ecological health. *In* Constanza, R Norton, B. and Haskell, B. (*eds*). *Ecosystem Health*: *New Goals for Environmental Management.* Island Press: Washington.

Hannon, B., 1999. Indicators of economic and ecological health. *In* Van den Bergh, J. C. J. M. (*ed*). *Handbook of Environmental and Resource Economics*. Edward Elgar: Cheltenham.

Hettinger, N. and Throop, B., 1999. Refocusing Ecocentrism: De-emphasising stability and defending wildness. *Environmental Ethics*, 21:3-21.

Hirschleifer, J., 1982. Evolutionary models in economics and law: Co-operation versus conflict strategies. *In* Lerbe, R. O. and Rubin, P. H. (*eds*). *Research in Law and Economics*, 4: 1-60.

Hobbs, R. J. and Norton, D. A., 1996. Towards a conceptual framework for restoration ecology. *Restoration Ecology*, 4: 93-110.

Hoehn, J. P., Lupi, F. and Kaplowitz, M. D., 2003. Untying a Lancasterian bundle: valuing ecosystems and ecosystem services for wetland mitigation. *Journal of Environmental Management*, 68: 263-272.

Holling, C. S., 1973. Resilience and stability of ecological systems. *Annual Review of Ecological Systems*, 4: 1-23.

Holling, C. S., 1978. *Adaptive Environmental Assessment and Management*. John Wiley: New York.

Holling, C. S., 1986. Resilience of ecosystems: Local surprise and global change. *In* Clark, E. C. and Munn, R. E. (*eds*). *Development of the Biosphere*. Cambridge University Press, Cambridge.

Holling, C. S., Schindler, D. W., Walker, B. H. and Roughgarden, J., 1995. Biodiversity in the functioning of ecosystems: An ecological synthesis. *In* Perrings, C. A., Maler, K. G., Folke, C., Holling, C. S. and Jansson, B. O. (*eds*). *Biodiversity Loss*: *Ecological and Economic Issues*. Cambridge University Press: Cambridge.

Holling, C. S. and Sanderson, S., 1996. Dynamics of (dis)harmony in ecological and social systems. *In* Hanna, S. S., Folke, C. and Maler, K. G. (*eds*). *Rights to Nature*: *Ecological, Economic, Cultural, and Political Principles of Institutions for the Environment*. Island Press: Washington.

Holling, C. S., 2001. Understanding the complexity of economic, ecological, and social systems. *Ecosystems*, 4: 390-405.

Holling, C. S. and Gunderson, L. H., 2002. Resilience and adaptive cycles. *In* Gunderson, L. H. and Holling, C. S. (*eds*.). *Panarchy*: *Understanding Transformations in Human and Natural Systems*. Island Press: Washington.

Hunter, M., 1988. Benchmarks for managing ecosystems: Are human activities natural? *Conservation Biology*, 10: 695-697.

Ives, A. R., 1995. Measuring resilience in stochastic systems. *Ecological Monograph*, 65: 217-233.

Jorgensen, S. E., 1997. *Integration of Ecosystem Theories*: *A Pattern*. Kluwer Academic: Dordrecht.

Kamien, M. I. and Schwartz, N. L., 1991. *Dynamic Optimization*: *The Calculus of Variations and Optimal Control in Economics and Management*. North-Holland: Amsterdam.

Kapustka, L. A. and Landis, W. G., 1998. Ecology: The science versus the myth. *Human and Ecological Risk Assessment*, 4: 829-838.

Karr, J. R., 1981. Assessment of biotic integrity using fish communities. *Fisheries*, 6: 21-27.

Karr, J. R., Fausch, K. D., Angermeier, P. L., Yant, P. R. and Schlosser, I. J., 1986. *Assessing biological integrity in running waters*: *A method and its rationale*. Illinois Natural History Survey Special Publication 5.

Karr, J. R., 1996. Ecological integrity and ecological health are not the same. *In* Schulze, P. C., Frosch, R. and Risser, P. (*eds*.). *Engineering within Ecological Constraints*. National Academy of Engineering: Washington.

Karr, J. R. and Chu, E. W., 1999. *Restoring life in running waters*: *Better biological monitoring*. Island Press: Washington.

Kauffman, 1993. *Origins of Order*: *Self-Organisation and selection in Evolution*. Oxford University Press: Oxford.

Kay, J. J., 1984. *Self-Organisation in Living Systems*. Systems Design Engineering Thesis, University of Waterloo: Ontario.

Kay, J. J., 1991. A nonequilibrium thermodynamic framework for discussing ecosystem integrity. *Environmental Management*, 15: 483-495.

Kay, J. J., 1993. On the nature of ecological integrity: Some closing comments. *In* Woodley, S., Kay, J. J. and Francis, G. (*eds*.). *Ecological Integrity and the Management of Ecosystems*. St. Lucie Press: Delray.

Kay, J. J. and Schneider, E., 1994. Embracing complexity, the challenge of the ecosystem approach. *Alternatives*, 20: 32-39.

Kay, J. J., Regier, H., Boyle, M. and Francis, G., 1997. An ecosystem approach for sustainability: Addressing the challenge of complexity. *Futures*, 31: 721-742.

Kay, J. J. and Regier, H. A., 2000. Uncertainty, complexity, and ecological integrity: Insights from an ecosystem approach. *In* Crabbe, P. (*ed*.). *Implementing Ecological Integrity*. Kluwer Academic.

Kay, J. J., Allen, T., Fraser, R., Luvall, J. C. and Ulanowicz, R., 2001. Can we use energy based indicators to characterize and measure the status of ecosystems, human, disturbed and natural? *Proceedings of the international workshop*: *Advances in Energy Studies: Exploring Supplies, Constraints and Strategies*, Porto Venere.

Kimmins, J. P., 1996. The health and integrity of forest ecosystems: Are they threatened by forestry? *Ecosystem Health*, 2: 5-18.

Kuhn, T., 1962. *The Structure of Scientific Revolutions.* University of Chicago Press: Chicago.

Lackey, R. T., 2001. Values, policy and ecosystem health. *BioScience*, 51: 437-443.

Lancaster, K., 1971. *Consumer Demand*. *A New Approach*. Columbia University Press: New York.

Lancaster, J., 2000. The ridiculous notion of assessing ecosystem health and identifying the useful concepts underneath. *Human and Ecological Risk Assessment*, 6: 213-222.

Lele, S. and Norgaard, R. B., 1996. Sustainability and the scientist's burden. *Conservation Biology*, 10: 354-365.

Lempert, R. J., Schlesinger, M. E. and Bankes, S. C., 1996. When we don't know the costs or the benefits: Adaptive strategies for abating climate change. *Climatic Change*, 33: 235-274.

Leopold, A., 1939. A biotic view of land. *Journal of Forestry*, 37: 727-730.

Leopold, A., 1939*b*. The farmer as a conservationist. *American Forests*, 45: 294-299.

Leopold, A., 1941. Wilderness as a land laboratory. *Living Wilderness*, 6: 3.

Levin, S. A., 1995. *Biodiversity*: *Interfacing Populations and Ecosystems*. Kyoto University Press: Kyoto.

Levin, S. A., 1999. *Fragile Dominion*. Perseus Publishing: Reading.

Lewin, R., 1992. *Complexity*: *Life at the Edge of Chaos*. MacMillan Publishing: New York.

Limburg, K. E., O'Neill, R. V., Costanza, R. and Farber, S., 2002. Complex systems and valuation. *Ecological Economics*, 41: 409-420.

Lockwood, J. L. and Pimm, S. L., 2001. When does restoration succeed? *In* Weiher, E (*ed*.). *Ecological Assembly Rules*. Cambridge University Press: Cambridge.

Lockwood, M., 1996. End value, evaluation, and natural systems. *Environmental Ethics*, 18: 265-278.

Loreau, M., Naeem, S., Bengtsson, J., Grime, J. P. Hector, A., Hooper, D. U. Huston, M. A., Raffaelli, D., Schmind, B., Tilman, D. and Wardle, D. A., 1999. Biodiversity and ecosystem functioning: Current knowledge and future challenges. *Science*, 294: 804-808.

Lotka, A. J., 1925. *Elements of Physical Biology*. Williams & Wilkins: Baltimore.

Loucks, O., 2000. Pattern of forest integrity in the eastern U. S. and Canada: Measuring loss and recovery. *In* Pimentel, D., Westra, L. and Noss, R. (*eds*.). *Ecological Integrity*: *Integrating Environment*, *Conservation and Health*. Island Press: Washington.

Ludwig, D., Hilborn, R. and Walters, C., 1993. Uncertainty, resource exploitation and conservation: Lessons from history. *Science*, 260: 17-36.

Ludwig, D., Walker, B. and Holling, C. S., 1997. Sustainability, stability and resilience. *Conservation Ecology*, 1: 1-23.

MacArthur, R. H., 1955. Fluctuations of animal populations, and a measure of community stability. *Ecology*, 36: 533-536.

MacArthur, R. H., 1972. Geographical Ecology. Harper & Row: New York.

Mageau, M. T., Costanza, R. and Ulanowicz, R. E., 1995. The development and initial testing of a quantitative assessment of ecosystem health. *Ecosystem Health*, 1: 201-213.

Marhsall, A., 1949. *The Principles of Economics*. Macmillan: London.

May, R. M., 1973. *Stability and Complexity in Model Ecosystems*. Princeton University Press: Princeton.

McGlone, M. S., 1983. Polynesian deforestation of New Zealand: A preliminary synthesis. *Archaeology in Oceania*, 18: 11-25.

McGlone, M. S., 1989. The Polynesian settlement of New Zealand in relation to environmental and biotic changes. *New Zealand Journal of Ecology*, 12: 115-119.

Meyer, J. L., 1997. Stream health: Incorporating the human dimension to advance stream ecology. *Journal of the North American Benthological Society*, 16: 439-447.

Mills, L. S., Soule, M. E. and Doak, D. F., 1993. The keystone species concept in ecology and conservation, *BioScience*, 43: 219-224.

Mirowski, P., 1989. The probabilistic counter-revolution or how stochastic concepts came to neoclassical theory. *Oxford Economic Papers*, 41: 217-235.

Moore, G. E., 1903. *Principia Ethica*. Cambridge University Press: Cambridge.

Muller, F., 1998. Gradients in ecological systems. *Ecological Modeling*, 108: 3-21.

Naemm, S., Thompson, L. J., Lawler, S. P. Lawton, J. H. and Woodfin, R. M., 1994. Declining biodiversity can alter the performance of ecosystems. *Nature*, 368: 734-737.

Neubert, M. G. and Caswell, H., 1997. Alternatives to resilience for measuring the responses of ecological-systems to perturbation. *Ecology*, 78: 653-665.

Nielsen, S. N. and Ulanowicz, R. E., 2000. On the consistency between thermodynamical and network approaches to ecosystems. *Ecological Modelling*, 132: 23-31.

Norgaard, R. B., 1984. Coevolutionary development potential. *Land Economics*, 60: 160-173.

Norgaard, R. B., 1994. *Development Betrayed*: *The End of Progress and a Co-evolutionary Revision of the Future*. Routledge: London.

Noss, R. F., 1990. Can we maintain our biological and ecological integrity? *Conservation Biology*, 4: 241-243.

Nunes, P. A. L. D., van den Bergh, J. C. J. M. and Nijkamp, P., 2003. *The Ecological Economics of Biodiversity*: *Methods and Applications*. Edward Elgar: Cheltenham.

Odum, E. P. 1969. The strategy of ecosystem development. *Science*, 164: 262-270.

Odum, H. T., 1993. *Ecological and General Systems*: *An Introduction to Systems Ecology*. University of Colorado Press: Niwot.

Odum, H. T., 1996. *Environmental Accounting*: *Emergy and Environmental Decision Making*. Wiley: New York.

Oechel, W. C., Cowles, S., Grulke, N., Hastings, J., Lawrence, B., Prudhomme, T., Riechers, G., Strain, B., Tissue, D. and Vourltis, G., 1994. Transient nature of CO_2 fertilization in Arctic tundra. *Nature*, 371: 500-503.

O'Laughlin, J., Livingston, R. L., Their, R., Thornton, J., Toweill, D. E. and Morelan, L., 1994. Defining and measuring forest health. *Journal of Sustainable Forestry*, 2: 65-85.

Olmsted, J., 1988. Observations on evolution. *In* Weber, B. H., Depew, D. J. and Smith, J. D. (*eds*.). *Entropy, information and Evolution*: *New Perspectives on Physical and Biological Evolution*. MIT Press: Cambridge.

O'Neill, R.V., DeAngelis, D.L., Waide, J.B., and Allen, T.F.H., 1986. *A Hierarchical Concept of Ecosystems*. Princeton University Press: Princeton.

Pahl-Wostl, C., 1995. *The Dynamic Nature of Ecosystems*: *Chaos and Order Entwined*. John Wiley: New York.

Paine, R. T., 1969. A note on trophic complexity and community stability. *American Naturalist*, 103: 91-93.

Pareto, V., 1897. *Cours d'Economie Politique*. Lausanne.

Perrings, C. and Pearce, D., 1994. Threshold effects and incentives for the conservation of biodiversity. *Environmental and Resource Economics*, 4: 13-28.

Perrings, C. A., Maler, K-G., Folke, C., Holling, C. S. and Jansson, B-O., 1995. *Biodiversity Loss*: *Ecological and Economic Issues*. Cambridge University Press: Cambridge.

Peterson, G. D., Allen, C. R. and Holling, C. S., 1998. Ecological resilience, biodiversity and scale. *Ecosystems*, 1: 6-18.

Peterson, G. D. and Pritchard, R., 2002. Resilience in social-ecological systems: A working hypothesis for a participatory approach. *Conservation Ecology*, 6: 14-24.

Pimm, S. L., 1984. The complexity and stability of ecosystems. *Nature*, 307: 321-326.

Pimm, S. L., 1991. *The Balance of Nature*? *Ecological Issues in the Conservation of Species and Communities*. The University of Chicago Press: Chicago.

Potts, J., 2000. *The New Evolutionary Microeconomics*: *Complexity, Competence and Adaptive Behaviour*. Edward Elgar: Cheltenham.

Power, M. E., Tilman, D., Estes, J. A., Menge, B. A., Bond, W. J., Mills, L. S., Daily, G., Castilla, J. C>, Lubchenko, J. and Paine, R. T., 1996. Challenges in the quest for keystones. *BioScience*, 46: 609-620.

Prigogine, I., 1980. *From Being to Becoming*. Freeman: San Francisco.

Puu, T., 2000. *Attractors, Bifurcations, and Chaos*: *Nonlinear Phenomena in Economics*. Springer: Berlin.

Randall, A., 1988. What mainstream economists have to say about the value of biodiversity. *In* Wilson, E. O. (*ed*.) *Biodiversity*. National Academy Press: Washington.

Rapport, D. J., 1989. What constitutes ecosystem health? *Perspectives in Biology and Medicine*, 33: 120-132.

Rapport, D. J., Costanza, R. and McMichael, A. J., 1998. Assessing ecosystem health. *Trends in Ecology and Evolution*, 13: 397-402.

Reid, W. V., 1994. Setting objectives for conservation evaluation. *In* Forey, P. I., Humphries, C. J. and Vane-Wright, R. I. (*eds*.). *Systematics and Conservation Evaluation*. Clarendon Press: Oxford.

Regier, H. A., 1993. The notion of natural and cultural integrity. *In* Woodley, S. J., Kay, J. J. and Francis, G. (*eds*.). *Ecological Integrity and the Management of Ecosystems*. Saint Lucie Press: Delray Beach.

Robinson, J., 1962. *Economic Philosophy*. Penguin: New York.

Rozdilsky, I. D. and Stone, L. S., 2001. Complexity can enhance stability in competitive systems. *Ecology Letters*, 4: 397-400.

Ryder, R. A., 1990. Ecosystem health, a human perception: Definition, detection, and the dichotomous key. *Journal of Great Lakes and Reserves*, 16: 619-624.

Sagoff, M., 1995. The value of integrity. *In* Westra, L. and Lemons, J. (*eds*.). *Perspectives on Ecological Integrity*. Kluwer Academic: Dordrecht.

Schaeffer, D. J., Henricks, E. E. and Kerster, H. W., 1988. Ecosystem health: Measuring ecosystem health. *Environmental Management*, 12: 445-455.

Schindler, D. W., 1990. Natural and anthropogenically imposed limitations to biotic richness in freshwaters. *In* Woodwell, G. (*ed*.). *The Earth in Transition*: *Patterns and Processes of Biotic Impoverishment*. Cambridge University Press: Cambridge.

Schneider, E. D. and Kay, J. J., 1994. Complexity and Thermodynamics: Towards a New Ecology. *Futures*, 24: 626-647.

Schneider, E. D. and Kay, J. J., 1995. Order from disorder: The thermodynamics of complexity in biology. *In* Murphy, M. P. and O'Neill, L. A. J. (*eds*.). *What is Life*: *The Next Fifty Years*. Cambridge University Press: Cambridge.

Schrodinger, E., 1944. *What is Life?* Cambridge University Press: Cambridge.

Schumpeter, J. A., 1954. *History of Economic Analysis*. Oxford University Press: New York.

Schumpeter, J. A., 1964. *Business Cycles*: *A Theoretical, Historical and Statistical Analysis of the Capitalist Process*. McGraw-Hill: New York.

Schuster, P. and Sigmund, K., 1980. Self-organisation of biological macromolecules and evolutionary stable strategies. *In* Haken, H. (*ed*.). *Dynamics of Synergetic Systems*. Springer-Verlag: London.

Scrimgeour, G. J. and Wicklum, D., 1996. Aquatic ecosystem health and integrity: Problems and potential solutions. *Journal of the North American Benthological Society*, 15: 254-261.

Simberloff, D., 1997. Flagships, umbrellas, and keystones: Is single-species management passé in the landscape era? *Biological Conservation*, 83: 247-257.

Smith, A., 1776. *The Wealth of Nations*. Methuen: London.

Solow, R., 1973. Is the end of the world at hand? *Challenge*, March: 39-50.

Steedman, R. J., 1994. Ecosystem health as a management goal. *Journal of the North American Benthological Society*, 13: 605-610.

Stephens, T., Brown, D. and Thornley, N., 2002. *Measuring Conservation Achievement*: *Application of the Concepts over the Twizel Area*. Science for Conservation 200, Department of Conservation: Wellington.

Swanson, F. J., Jones, J. A., Wallin, D. O. and Cissel, J. H., 1993. Natural variability – implications for ecosystem management. *In* U. S Forest Service (*eds*.). *Eastside Forest Ecosystem Health Assessment*: Ecosystem Management. U. S. Forest Service: Portland.

Swanson, T. M., 1995. *The Economics and Ecology of Biodiversity Decline*. Cambridge University Press: Cambridge.

Tansley, A. G. 1920. The classification of vegetation and the concept of development. *Journal of Ecology*, 8: 118-149.

Thom, R., 1975. *Structural Stability and Morphogenesis*. Benjamin: Reading.

Thompson, J., 2000. Environment as cultural heritage. *Environmental Ethics*, 22: 241-258.

Tilman, D. and Downing, J. A., 1994. Biodiversity and stability in grasslands. *Nature*, 367: 363-365.

Tracy, C. R. and Brussard, P. F., 1994. Preserving biodiversity: Species in landscapes. *Ecological Applications*, 42: 205-207.

Ulanowicz, R. E., 1980. An hypothesis on the development of natural communities. *Journal of Theoretical Biology*, 85: 223-245.

Ulanowicz, R. E., 1986. *Growth and Development*: *Ecosystems Phenomenology*. Springer-Verlag: New York.

Ulanowicz, R. E., 1997. *Ecology*: *The Ascendant Perspective*. Columbia University Press : New York.

Van der Leeuw, S. E., 2000. Land degradation as a socio-natural process. *In* McIntosh, R. J., Tainter, J. A. and McIntosh, S. K. (*eds*.). The Way the Wind Blows: Climate, History and Human Action. Columbia University Press: New York.

Van Kooten, G. C. and Bulte, E. H., 1999. *The Economics of Nature*: *Managing Biological Assets*. Blackwell: Oxford.

Varian, H. R., 1992. *Microeconomic Analysis*. W. W. Norton: New York.

Veblen, T., 1898. Why is economics not an evolutionary science? *Quarterly Journal of Economics*, 12: 373-397.

Vitousek, P. M., 1990. Biological invasions and ecosystem processes: Towards an integration of population biology and ecosystem studies. *Oikos*, 57: 7-13.

Vitousek, P. M., Mooney, H. A., Lubchenco, J. and Milillo, J. M., 1997. Human domination of earth's ecosystems. *Science*, 277: 494-499.

Von Bertalanffy, L., 1968. *General System Theory*: *Foundations, Development, Applications*. George Braziller: New York.

Walker, B. H., Ludwig, D., Holling, C. S. and Peterman, R. M., 1981. *Journal of Ecology*, 69: 473-498.

Walker, B. H., 1992. Biodiversity and ecological redundancy. *Conservation Biology*, 6: 18-23.

Walker, B. H., Langridge, J. L. and McFarlane, F., 1997. Resilience of an Australian savannah grassland to selective and non-selective perturbations. *Australian Journal of Ecology*, 22: 125-135.

Walker, J. L. and Boyer, W. D., 1995. An ecological model and information needs assessment for longleaf pine ecosystem restoration. *In* U. S. Forest Service (*eds*.). *Silviculture*: *From the Cradle of Forestry to Ecosystem Management*. General Technical Report SE-88: North Carolina.

Walters, C. J., 1986. *The Adaptive Management of Renewable Resources*. MacMillan Press: New York.

Wassenaar, T. D. and Ferreira, S. M., 2002. *Measuring conservation outcomes for depleted biological assets*: *Convergence theory and its applicability to New Zealand*. DOC Science Internal Series 38, Department of Conservation: Wellington.

Weiderholm, T., 1980. *Use of eutrophication in Australian inland waters*. Australian Water Resource Council Technical Paper 15, Australian Government Publishing Service: Canberra.

Wicklum, D. and Davies, R., 1995. Ecosystem Health and Integrity. *Canadian Journal of Botany*, 73: 997-1000.

Wolfgram, S., 1984. Cellular automata as models of complexity. *Nature*, 311: 419-424.

Worster, D., 1979. *Nature's Economy*: *The Roots of Ecology*. Anchor Press: New York.

Wulff, F., Field, J. G. and Mann, K. H., 1989. *Network Analysis of Marine Ecosystems*: *Methods and Applications*. Springer-Verlag: Heidelberg.