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Why is achieving good ecological outcomes in rivers so difficult?

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SUMMARY

1. Considerable evidence from around the world shows that achieving good ecological outcomes in rivers from programmes of measures in catchments is difficult. There are a number of reasons for this, which we discuss, but here we focus primarily on the question, 'Is the knowledge base adequate?'
2. We develop further the thesis that catchments and receiving waters are truly complex systems in which there are fundamental limits to knowledge. Our sampling and data analysis practices come with strong biases and inbuilt 'framing' assumptions about the nature and values of ecosystem processes that underestimate complexity and uncertainty.
3. If we reframe our assumptions to think of the problem in terms of the properties of complex systems, then we can rethink our attitudes to uncertainty, causal thickets (multiple stressors) and cross-scale effects, and we can begin to develop new definitions of what constitutes a 'good' ecological outcome. Dealing with inherent variability in data then becomes less of a problem with controlling 'noise' and more of a problem of understanding system dynamics. The presence of adaptive dynamics and self-organisation in complex systems means that uncertainties will always be large and knowledge will be partial and that such systems are fundamentally not computable.
4. We show that small-scale 'noise' in ecosystems is an inherent property of non-equilibrium systems with predominant advection, reaction–diffusion dynamics. Flow paths in catchments and the dynamics of receiving waters have fractal properties. Fractal dynamics indicate that multiple, cross-scale interactions are a characteristic of these systems. Small-scale connectivity is an important (and, from a management point of view, underused) aspect of pattern and process in such systems.
5. In an environment of such complexity, models will be flawed and predictions uncertain. It is therefore necessary to develop new indicators of connectivity and ecological complexity so that indicators of system-level progress may be found to assist with an improved process of adaptive management that is trend-orientated as well as outcome-orientated.

Keywords: catchments, complexity, restoration, rivers, uncertainty

Missed opportunities for integrating land and water management for ecological gain

In this paper, we focus primarily on the connections between land use, water quality and the ecological status of rivers – and the success, or otherwise, of attempts to restore the ecological status of rivers through the implementation of programmes of measures in catchments: management actions designed to produce improved ecological outcomes. From the results of numerous restoration attempts around the world, it is now becoming clear that success in this respect is not easy to come by. We do not always seem to achieve the desired outcomes, for a number of

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reasons. Inter alia, the problems seem to relate to the following: (i) there is less predictability and more uncertainty in the ecological response than we had anticipated, (ii) follow-up monitoring programmes are often inadequate and (iii) programmes of measures do not always display sufficient 'joined-up thinking' so that the original outcomes were not clearly defined, the knowledge base and hence the design was poor and/or there was insufficient effort or community engagement to ensure success. Problems can certainly arise when management actions are not supported by a sufficient knowledge basis (Rogers & Biggs, 1999; Biggs & Rogers, 2003).

Catchments are complex. Cross-scale interactions in landscapes really matter (Green & Sadedin, 2005) but there is a tendency to discount the scale and degree of natural pattern, process and variability (and the emergence of large-scale properties from fine-scale interactions) and instead – through models – try to develop generic thinking to overlay the natural patterns and processes of the biosphere with the convenience of human institutional, social and economic scales (Harris, 2007). There is much 'particularity of place' and much small-scale pattern and process in soils and terrestrial ecosystems that confounds generic approaches but could be captured by appropriate 'respect' for uncertainty in ecological response. Catchments differ in their connectivity, both between in-stream components and between critical source areas in the catchment and the channel (Reaney *et al.*, 2010). We generally fail to use this connectivity to help focus adaptive management towards potential positive outcomes at the catchment scale.

Rivers are themselves self-organised structures with a naturally fine balance between flow, erosion and sedimentation (Fagherazzi, 2008). Rivers are well known to exhibit fractal structural properties (e.g. De Bartolo, Veltri & Primavera, 2006; Saa *et al.*, 2007; Rodriguez-Iturbe *et al.*, 2009) and are examples of the long-term working out of power law distributions of things like bank failures and meanders (Fonstad & Marcus, 2003; Frascati & Lanzoni,2010). So changes in river flows, geomorphology and catchment drainage can have short- and long-term consequences – even out to centuries; and in many parts of the world, we have also been modifying (and 'improving') rivers for centuries (see e.g. Walter & Merritts, 2008). So we have yet to see the final outcomes of many of our actions. Against this background, the river ecology is also a spatially distributed, multi-scale ecosystem in which dispersal, connectivity and network structures are important (Lowe, Likens & Power, 2006). One aspect in particular – that of the potential for lengthy time lags between management action and ecological result – is rarely acknowledged.

Groundwater flow paths – which also connect the catchment to the stream – are complex (Fuchs *et al.*, 2009) and important (Fisher, Sponseller & Heffernan, 2004), and fine-scale pattern and process are manifest as large-scale landscape properties (Ryan, Ludwig & McAlpine, 2007), often with significant time lags. Calculations of catchment loads of key nutrients and materials, which might be expected to have ecological impacts (and therefore be the subject of management measures, e.g. C, N, P and sediments), are not simple because of the effects of preceding rainfall, catchment wetness, varying flow paths and the balance of point and diffuse sources arising from differing critical source areas (e.g. Heathwaite & Dils, 2000; Bowes *et al.*, 2010; Jarvie *et al.*, 2010). So there are many fractal and self-organised catchment and riverine biophysical properties that involve the long-term working out of cross-scale interactions.

Relationships between land use and the ecological state of rivers appear to be quite weak. For example, a recent analysis of the ecological condition of 400 headwater streams in the U.K. (assessed by both macroinvertebrate and macrophyte abundance and biodiversity, Murphy, 2010) showed that adjacent land use accounted for 5–25% of the observed variances. This appears to be a common observation. So we have low power when we change catchment land use: major changes are probably required to produce observable effects on river condition, and response may not be immediate. Further problems arise from the choice of assessment tools and descriptors of ecological condition (Noges *et al.*, 2009; Downes, 2010), from nonlinear and threshold responses (Allan, 2004) and from sampling schemes and data sources, which do not describe the true scales of pattern and process in catchments (Harris & Heathwaite, 2005; Heathwaite, 2010).

Because of poor follow-up monitoring after river restoration efforts, it is actually quite hard to demonstrate that anything has been achieved, let alone good results (Bernhardt *et al.*, 2005; Palmer *et al.*, 2007). In addition, institutional issues around data archiving make it hard to sustain long-term monitoring in a research and policy environment dominated by a focus on short-term programmes (Brooks & Lake, 2007).

A recent review of over 2000 river restoration projects showed that success rates are low, and there is little scientific assessment or monitoring of these projects (Brooks & Lake, 2007). In Brooks & Lake's (2007) analysis, only about 14% of the management interventions assessed were monitored properly. In the USA, \$14–15 billion has been spent on river restoration projects in the last 20 years, and only 10% of these projects were properly monitored (Bernhardt *et al.*, 2005). In Australia, a meta-analysis of more than 550 data sets that attempted to demonstrate ecological benefits from the restoration of environmental flows in rivers showed that only about 30 data sets were useful (Overton *et al.*, 2010).

So, overall, success rates are low. Moss (1999) described the most recent age of freshwater conservation as 'sans teeth' and 'as the triumph of hope over experience'. It is not always clear what we are trying to restore to what, or what the beginning and end points actually are (or were). Success rates are not high, and unaltered baselines do not usually exist (Duarte *et al.*, 2009). Certainly, all the evidence points to the fact that some major investments in integrated water resource management and agri-environment schemes have delivered little (Kleijn *et al.*, 2006; Kay, Edwards & Foulger, 2009). In the same vein, Moss (2007, 2008) has produced some of the most cogent criticisms of the EU Water Framework Directive relying, as it does, on measuring symptoms and 'simple taxonomic indices as measures of (ecological) quality' (Moss, 2007). Further, management directives, like the Water Framework Directive, rely on some strong assumptions about our ability to unequivocally link programmes of measures in catchments to ecological outcomes. As Downes (2010) has pointed out, the knowledge base is often not adequate to the task because of poor sample designs and the use of assessment data as an input to predictive models. So, for a range of reasons to do with sampling protocols and models, data adequacy and institutional factors (including assumptions about knowledge, predictability and uncertainty), achieving good ecological outcomes (or demonstrating that this is so) is proving to be a challenge.

Yet, the regulation and legislation incorporated in policy assume an ability to predict the likely outcomes of management actions and the ability to implement measures to produce the desired change in, for example, ecological condition; it is also assumed that uncertainties are small. Above all, when attempting to restore rivers to 'good' ecological condition, we make numerous assumptions about the linkages between programmes of measures implemented in the catchment and outcomes in the water. We assume, as Hynes (1975) did, that 'the valley rules the stream' (but we perhaps forget that he also said that each stream is an individual). Because of the complexity of the properties of catchments that have recently been revealed by detailed monitoring (e.g. fractal flow paths connecting sources in catchments to the river, Kirchner, Feng & Neal, 2001; and apparently self-organised properties, Harris & Heathwaite, 2005; and references therein), programmes of measures carried out in catchments do not usually take into account the complexities of pattern and process either in the rivers themselves or in the fundamental soil and landscape properties (Harris, 2002a,b, 2003, 2007).

We have known for some time that attempts at ecological restoration do not usually bring the ecosystem back to the same place as it once was. Ecosystems are always in flux, so restoration trajectories usually differ from the route taken during degradation (Jeppesen *et al.*, 2005; Moss *et al.*, 2005; Kondolf *et al.*, 2006; Suding & Gross, 2006; Duarte *et al.*, 2009). This is not to say that government agencies, national and international NGOs, catchment management authorities, river trusts and community-based groups of various kinds have not made many experiments in local, place-specific conservation, management and restoration. Local successes have been recorded, but the larger-scale, and global, picture remains negative (GBO, 2010). Restoration ecology is, to some, a troubled discipline, and it is clear that we live in an era in which many novel ecosystems are emerging with new species combinations arising from deliberate or inadvertent human action (Hobbs *et al.*, 2006).

EU stewardship programmes and agri-environment schemes are perhaps a missed opportunity in terms of linking changes in land management to anticipated outcomes in terms of freshwater quality. Their focus has been largely on terrestrial habitats and not on an integrated view of catchment outcomes that would encompass freshwater quality. A review of the terrestrial outcomes of the EU agri-environment schemes showed, at best, equivocal results (Kleijn

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et al., 2006). Further, for many agricultural stewardship measures, there is little or no evidence of beneficial impacts on water quality; mark you, these programmes were designed to preserve assets like biodiversity and heritage values rather than hydrological or biogeochemical properties of catchments, so it is not likely that they will be successful in this regard. However, few (or no) studies have been carried out to quantify the impact of agricultural stewardship measures on water quality at the catchment scale (Kay *et al.*, 2009). These programmes could do a better job improving water quality if programmes of measures were better strategically targeted to achieve multiple outcomes (including water quality) by, for example, ensuring that areas set aside for biodiversity conservation also target critical source areas for diffuse pollution in catchments. 'Joined-up thinking' would seek multiple beneficial outcomes (biophysical as well as biodiversity) from single measures.

There is therefore much resulting uncertainty in measures, indices, data, models and outcomes. Newig, Pahl-Wostl & Sigel (2005) differentiated between normative uncertainty – or decision uncertainty (doubt as to what to do and how to do it) – and informational uncertainty (limited knowledge or factual uncertainty). As Fig. 1 shows, it is necessary to

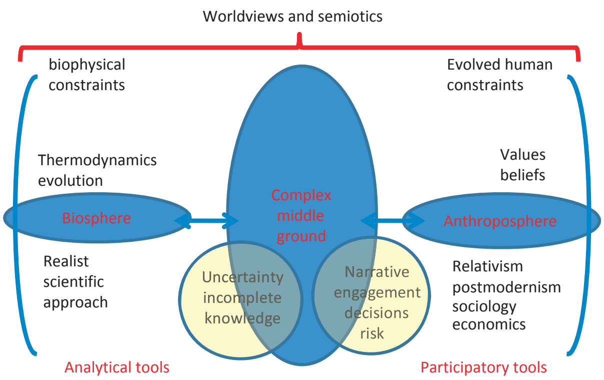

Fig. 1 The desired outcome, that of achieving improved ecological condition in receiving waters, relies on the interaction of the biosphere (and its ecology), and the anthroposphere (with its associated human institutions, beliefs and values). Both spheres are constrained: the biosphere by the evolved characteristics of life and by our partial knowledge of its complexity (informational uncertainty); the anthroposphere by the basic constraints of human values and beliefs about the world and by the cultures and institutions that articulate and implement them (normative uncertainty). In between them lies a complex middle ground where human society meets its environment.

consider both sides of the story if the problem of river restoration is to be tackled effectively. Over the last couple of decades, much effort has been made to reduce the uncertainty, but most of the effort in this regard has gone into engaging with stakeholders and the community in an effort to get good compliance with 'best management practices' and improve community engagement with programmes designed to improve catchment condition and reduce diffuse nutrient loads (the right-hand side of Fig. 1; Newig *et al.*, 2005). Much emphasis has been placed on participation tools, socio-economic considerations and community value and beliefs.

Less effort has been made to think seriously about experimental designs (Downes, 2010) and to obtain data that reduce fundamental uncertainties in the knowledge base (Heathwaite, 2010). So for a start, we wish to focus largely on the left side of the diagram and address the question of the adequacy of our knowledge about the world, our informational uncertainty and analytical tools. This will inevitably draw upon considerations of scientific beliefs, culture and human values arising from the right-hand side because these all strongly influence the design and execution of programmes of measures designed to restore rivers.

The way forward: reframing our assumptions and actions to encompass are more complex worldview (effectively making a 'complexity turn', Urry, 2005)

Over the last 30 years, Harris has developed arguments supporting a more complex view of the world than is normally espoused by ecologists. These arguments include Opinion papers in this journal (i.e. Harris, 1999a; and references therein, see also Harris & Heathwaite, 2005). Here, we explain how these arguments can be used to account for much of the uncertainty that we encounter and the inadequacies of many, if not most, of the designs and models used in projects such as river restoration. We draw here also on arguments developed in conference papers and technical reports: first in Harris (2002a,b, 2003) and then elaborated in Harris (2009).

When we conceptualise a problem to collect data and make predictions about the world, we design experiments and models with a host of inbuilt assumptions about what is important and what can be ignored, averaged out or otherwise dealt with statistically (Harris, 2007). There is a predominant 'framing' of environmental problems through equilibrium assumptions around frequentist statistics, Gaussian normality (or the ability to transform data to make it so) and the assumption of linearity, stationarity and 'white noise' variance distributions (Botkin, 1990). We make assumptions about the required data structures, about the necessary resolution of pattern and process and also about statistical distributions and moments (Harris, 2003, 2007). Our knowledge of ecological scale is limited by 'the arbitrary nature of observational-scale choice prevalent in today's literature' (Wheatley & Johnson, 2009). T.F.H Allen and co-workers have written extensively about the pervasive role of predisposition and designer choice in the process of model building and how choices based on culture and values are inevitably made (Giampietro, Allen & Mayumi, 2006; Zellner, Allen & Kesseboehmer, 2006). We make assumptions about what can be parameterised and what must be dealt with mechanistically (Beven, 2008a). In this way, we justify infrequent (fortnightly or less frequent) monitoring and assessment of river water quality, and less frequent monitoring of biota, accompanied by statistical treatment (averaging and normalisation) of the data (Harris, 2007).

One of the most basic problems in monitoring and managing catchments and the ecological response of rivers and receiving waters is that of linking cause and effect across scales in space and time. At any given point in a river catchment, impacts are both local and distant. Local linkages include *in situ* pattern and process plus nearby catchment influences, while the river provides an advective mechanism to bring distant land use change and other influences to bear. For this reason, there has been much reliance recently on hydrological and catchment modelling – the idea being to attempt to mechanistically link pattern and process across scales and to interpolate between the usually sparse data (Reaney *et al.*, 2010) – particularly sparse spatial data. The other reason for modelling is that of trying to unpick the interacting influences of multiple stressors: what Wimsatt (2007) has called causal thickets. At any given point, there are multiple causes and effects, often applying at overlapping scales in space and time, so that it is difficult to unequivocally determine precise linkages between management action and ecological outcome. Hence, we attempt to mechanistically combine our best knowledge and assumptions in models.

We must pay more attention to complexity and uncertainty in framing the problem of understanding the linkages between land use and the ecological condition of rivers. As we have previously argued (Harris, 1999a), reframing the question and thinking of a more complex reality (Urry, 2005) allow us to take a more nuanced view of uncertainty, non-stationarity, nonlinearity and non-normality; even the expectation of power law distributions and 'fat-tails' (Taleb, 2007) in which the probability of extreme events is much higher than we would expect from Gaussian distributions (Harris, 2009). In an era of climate change, 'stationarity is dead' (Milly *et al.*, 2008) and we need to critically review the inbuilt assumptions in many of our models, particularly as the frequencies of extreme events are undergoing rapid change. The properties of complex systems also include catastrophic failures, tipping points, hysteresis and emergence (Harris, 2007) – all of which are frequently observed in riverine systems (Allan, 2004). The recent Great Financial Crash has similarly focussed the minds of economists on increased uncertainty, the inadequacy of models and the 'framing' of economic problems and has placed much more emphasis on non-normal distributions and the finite risk of catastrophic events generated by the nonlinear behaviour of complex systems (Lo & Mueller, 2010). In essence, all data are partial and all models are uncertain (Beven, 2008a) because they are composed of data, assumptions and parameters that incorporate errors.

What the Great Financial Crash has done is to focus attention on epistemic or Knightian uncertainty (Lo & Mueller, 2010), i.e. the uncertainty contained in what we do not know and which lies outside our 'framing' assumptions (Donald Rumsfeld's famous comment about the importance of 'unknown unknowns' makes precisely this point). A more complex frame of reference brings into focus the inherently (and formally) non-computable and unpredictable properties of complex ecological and socio-economic systems (Urry, 2005; Carpenter *et al.*, 2009) and places emphasis on what is now called 'systemic risk' – risks arising from the incomplete knowledge of the coupled system and its dynamics (Deere-Birkbeck, 2009). Complexity places fundamental limits on what we can know and predict, but we should work with this rather than ignore it. A more complex frame of reference conditions us to expect surprises and constant change at all scales.

Carpenter et al. *(2009) have highlighted the two major shortcomings of the predominant ecological paradigm that frames the way we look at ecological quality.* First, the tendency to focus too much on the apparently computable aspects of environmental problems; even though environmental problems have much Knightian uncertainty (there is much that we do not know), complexity and emergence mean that some properties are formally not computable. Catchments and rivers contain much adaptive, interactive biology and ecology that cannot be ignored (Hauhs & Lange, 2008). So simulation models are overused, and uncertainties are underestimated (Beven, 2008a). We must not fall into the realist scientific trap of assuming that our models are true representations of reality; or, worse, assuming we can use models instead of data. Second, there is a strong 'group think' and belief in the dominant models so that inconsistent – and often complex – signals are filtered out, or, as Wynne (2005) has argued, actively denied (see also Harris, 2007, 2009).

Small-scale events are important. Just as the risk of system failure is underestimated by the usual philosophical 'framing' of ecological problems and 'causal thickets' are methodologically unresolvable (Wimsatt, 2007), we also underestimate the importance of small-scale events (and the likelihood of emergence) through assumptions about the ability to average out small-scale 'noise' (Harris, 2007). Models in particular, by defining pattern and process at certain (rather coarse) scales, make strong assumptions about what can be averaged out and what scales are important for understanding a particular problem (Harris, 2003). We tend to think in terms of kilometres and weeks rather than the required (much smaller, discrete) spatial and temporal scales that drive catchment and river ecology. Kareiva & Wennergren (1995) pointed out many years ago that spatially explicit ecological models (which are required in riverine systems) exhibit 'numerous surprises', and thresholds and have quite different properties from averaged, mean-field solutions (Harris, 2007). In addition, using ecological data originally collected for the purposes of ecological assessments does not satisfy the need for more rigorous experimental designs to resolve the links between actions and outcomes, and to calibrate and validate models (Downes, 2010). There is a reinforcement of design, resolution and philosophy when existing monitoring and assessment data are used as input to models used for predictive purposes; the philosophy, data and models specifically predetermine what is important and predictable at what scale. Even with the best community engagement, incomplete knowledge and pervasive uncertainty will defeat the best of intentions.

In what follows, we argue that reframing the questions, and using higher-resolution sampling and analysis (smaller spatial scales, higher temporal frequency), provides new insights into the complexity of pattern and process in catchments that will provide important contributions to successful management interventions.

Small-scale pattern and process in rivers and receiving waters

The data available on land and water processes point to more small-scale process-based variability in natural waters than we have hitherto anticipated. Natural waters are not well mixed in respect of biologically mediated factors. We now know that it requires at least daily data to resolve pattern and process in river water quality data (Butturini, 2005; Milne *et al.*, 2009). Harris & Heathwaite (2005) analysed daily river water quality time series and showed that there was information in what appeared, at first sight, to be 'noise' in those data. Similarly Jordan *et al.* (2007) showed that it required daily data from rivers to reveal the true (high frequency) scales of variability in pattern and process (see also Heathwaite, 2010). Harris (2002a, 2003) analysed weekly river data and showed that similar, apparently fractal, patterns of behaviour span scales of variability in river data from days to weeks.

Figure 2 shows one-minute resolution underway data (sampled by pumping from a moving vessel) from Longmore, Cowdell & Flint (1996) in Port Philip Bay, Victoria (corresponding to about 300-m spatial resolution), which show considerable non-random patchiness. This coastal embayment clearly has patches of water of varying chemical signatures at scales of a few hundred metres to a kilometre. This is consistent with previous observations of spatial distributions of biologically influenced substances in coastal waters where turbulence, physiology and growth interact. While univariate data series show fractal or multi-fractal properties in coastal waters (Seuront, Gentilhomme & Lagadeuc, 2002),

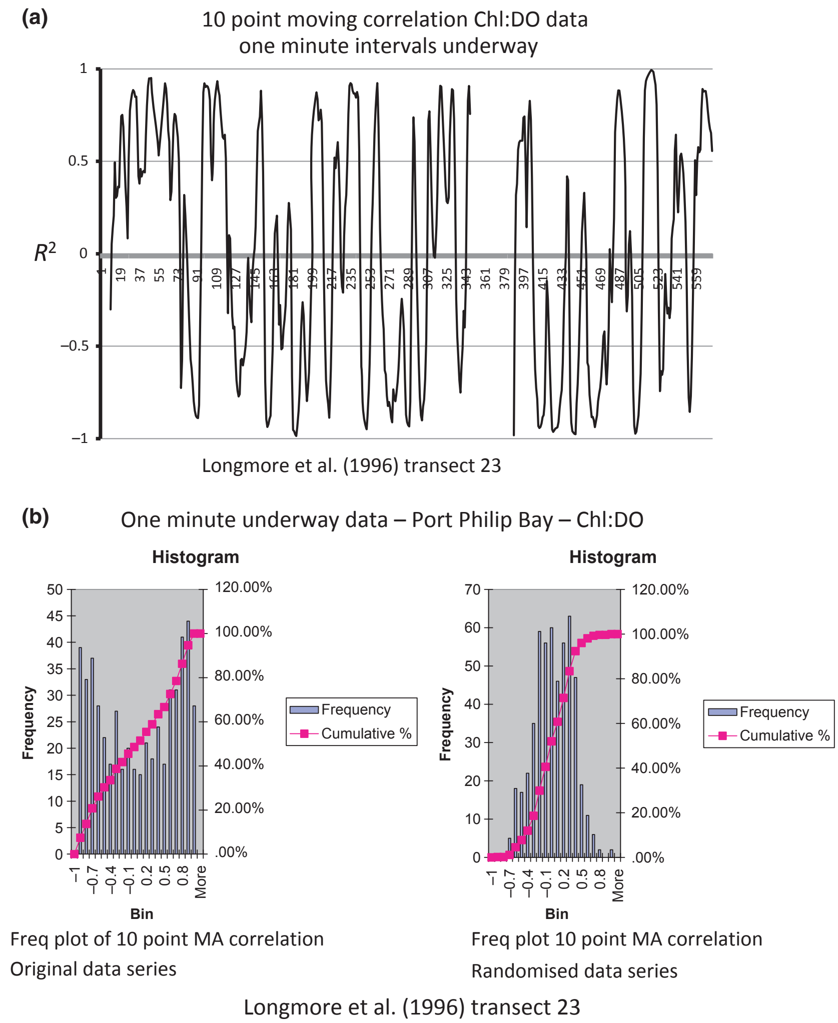

Fig. 2 (a) A 10-point moving correlation plot of 1-minute underway data from Port Philip Bay, Victoria. Data from Longmore *et al.* (1996). One-minute resolution represents about 300 m in the spatial dimension. (b) Histograms resulting from frequency plots of moving correlations, both raw data and after randomisation of the series. Clearly, the observed correlation distributions are decidedly non-random. Full methodology is provided in the study by Harris & Heathwaite (2005). (There was a gap in the sampling data between samples 345 and 390.)

multivariate data series show changing correlation patterns in all waterbodies and rivers (Harris & Trimbee, 1986; Harris, 1987; Harris & Heathwaite, 2005), indicating that the temporal evolution of the properties of small independent patches of water differs markedly.

We have known for more than 25 years that time-series analysis of correlations between water quality parameters and ecological variables in receiving waters shows epochs of changing signs and fluctuations in the underlying generating functions. These data are not stationary, and they are not normally distributed (Harris & Trimbee, 1986; Harris, 1987). More recently, using new analysis techniques, Milne *et al.* (2009) observed correlations of changing sign in river water quality time series using wavelet analysis. This, together with data from Scholefield *et al.* (2005), which show diurnal periodicities in river chemistry data from the River Taw in Devon, is a clear indication of much small-scale non-stationarity and changing relationships between variables.

Given the acknowledged presence of epochs of changing sign in ecological time series at scales from days to months, it is perhaps hardly surprising that attempts to derive through observation strong relationships between management actions in catchments and ecological outcomes in rivers are fraught with difficulty. Action does not always lead to identical outcomes – sometimes the opposite. Consequently, restoration trajectories may be surprisingly chaotic (Duarte *et al.*, 2009). We do know that the biota in separate catchments become adapted to the local biophysical drivers (Lytle, Bogan & Finn, 2008), and in some parts of the world, the biota in different catchments have been separated long enough to have different evolutionary histories (Overton *et al.*, 2010), so we can expect much uncertainty in the ecological response to management actions in catchments. This suggests that one size does not fit all, and as a result, models and management prescriptions may not be simply transportable from one place to another. This is sufficient to ensure that the relationship between action and outcome is weak and confounded, resulting in the observed lack of ecological outcomes.

In the context of this paper, a complex, non-equilibrium view of ecological outcomes 'reframes' the problem to one of path dependencies and trajectories over time; not the usual equilibrium view of the world. Path dependencies in biological responses ensure that even the purely hydrological properties of the catchment/riverine system is not capable of being treated as a purely physical problem (Hauhs & Lange, 2008). Interestingly, the data analysis shows that while the data reflect the strong effect of initial conditions (the varying sources of water in catchments) and of differing subsequent trajectories of development – both properties of complex systems – the data are clearly not chaotic (Fig. 2). There are patterns in the correlations; the initial conditions and the developing trajectories are constrained by the stoichiometry of biological process in the catchments and in the water. There is information in the 'noise' that is lost through infrequent sampling and data averaging (Harris & Heathwaite, 2005). The underlying stoichiometry is now known to be a basic property of the evolved molecular biology of living systems – particularly the physiology of the microscopic and the microbial denizens of soils, sediments, rivers and receiving waters (Sterner & Elser, 2002). The almost ubiquitous central tendency exhibited by these data (the Redfield Ratio) arises from the underlying molecular make-up of the biota. Molecular biology – which seems to have evolved only once – determines ecosystem-level responses (Elser *et al*., 2003; Falkowski, Fenchel & Delong, 2008).

Many years ago, Harris and co-workers analysed time-series data from lakes and showed that there is information in them which reflected a set of multiple influences: both catchment inputs and internal dynamics (Harris & Trimbee, 1986; Harris, 1987). By 'listening' to a catchment and river by sampling at any given point, we are essentially sampling from a set of constrained cross-scale dynamics representing a whole series of partly self-organised influences: climatological and hydrological drivers, changing connectivity, biogeochemical dynamics and exports from soils and catchment-based land uses (Milne *et al*., 2009). To 'listen' to the river, we need to be able to resolve the full spectrum of larger-scale advection and smaller-scale internal reaction–diffusion reactions. This would represent the full discrimination of the 'high-frequency wave of the future' (Kirchner *et al*., 2004) and allow the full orchestration of variability across scales to be understood. Few lake data sets have the required resolution (but see Harris, 1987), and almost no river data sets, partly because the investment is considerable and the approach needed goes against current paradigms of 'determinism'. Kirchner *et al*. (2001) have shown that catchment exports of conservative substances show fractal properties but, as far as we know, there are as yet no long-term, high-resolution spectra for reactive water quality parameters in the literature. Most of our water quality data from rivers, collected at widely spread locations and at weekly or fortnightly intervals, is severely aliased: that is, the data do not have sufficient temporal or spatial resolution to fully resolve the true scales of pattern and process. They do not resolve the small-scale dynamics present.

The only way to resolve these dynamics is to use a full particle-tracking computational fluid dynamics (CFD)-type model with high 3-D resolution and the inclusion of small-scale non-equilibrium temporally evolving patch dynamics (Fig. 2). While this type of modelling has been done for hydraulics and water quality in urban water pipe and treatment systems, and a 3-D CFD flow model has been used for the hydraulics of rivers (Rodriguez *et al*., 2004; Shen & Diplas, 2008), there do not appear to be any such coupled 3-D physics and water quality models for rivers. Essentially, we are advocating a high-resolution dynamic particle-tracking approach to the modelling of river water quality in which small-scale reaction–diffusion processes are embedded in a larger-scale turbulent mixing and advection framework (see the recent application to cloud physics in Bodenschatz *et al*., 2010). This is similar to the estuary modelling approach of Murray & Parslow (1999) but using higher-resolution coupled physical–biological modelling with multiple drivers. Such an approach is computationally intensive but necessary because we know that organisms in rivers are adapted and evolved to respond to and cope with the natural spectrum of variability encountered (Lytle *et al*., 2008). The ecological outcome is a nonlinear, adaptive and emergent, time-weighted function of the input spectrum and correlation structure (see Hauhs & Lange, 2008).

Reframing the question and taking a more complex approach focus attention on the natural balance of robustness and fragility exhibited by natural systems (Csete & Doyle, 2002), responding adaptively to the input spectrum and correlation structure of the ecological drivers. Systems which have adapted to the variability of the drivers and evolved a degree of highly optimised tolerance (HOT, Carlson & Doyle, 1999) are robust to anticipated disturbances but very fragile to unanticipated disturbances – without knowledge of the original spectrum and the multivariate structure of the natural disturbances or of the modified situation, it is little wonder that we are constantly surprised by the outcomes of anthropogenic change and fail in our restoration efforts. Complexity and robustness go hand in hand and are mutually reinforcing (Carlson & Doyle, 2002).

Links between catchments and receiving waters depend on scale

We know that the spectrum of ecological responses is constrained by evolutionary, stoichiometric and physiological constraints (Harris, 1999a,b, 2007). Despite the fractal and multivariate nature of the catchment drivers, we do know that there are transferable links between land use, catchment exports and the ecological response of receiving waters: rivers, lakes and estuaries. Vollenweider's (1968) OECD models are based on this fact. Yes, Vollenweider used annually averaged loads and parameters like plankton biomass (as chlorophyll), but the relationships developed have been widely used in lake restoration efforts. Indeed, Meeuwig & Peters (1996) showed that a simple direct empirical relationship between land use and lake chlorophyll worked marginally better than Vollenweider's (1975) model of in-lake total phosphorus coupled with the algal response. So there are regular responses of receiving waters to changing nutrient loads that are determined by the physiology and growth rates of the dominant organisms (Harris, 1994; Reynolds, Irish & Elliott, 2001; Webster & Harris, 2004). Vollenweider (1968) called this 'catchment physiology' (Harris, 2007). In other words, there are evolved constraints to the ecological drivers, many fundamentally based on the biochemistry and molecular biology of living organisms (Sterner & Elser, 2002). So while the relationship between aquatic biodiversity and land use is weak (Murphy, 2010), there are broad system-level constraints. As we have noted, even across diverse biota, stoichiometry influences C: N: P ratios and growth rates (Elser *et al*., 2003).

At large scales, there are predictable properties of rivers, dependent on land use and human populations in their catchments. Numerous statistical models of C, N and P exports from catchments have been constructed (e.g. Caraco *et al*., 2003; Smith *et al*., 2003; Maybeck *et al*., 2007) but, at the larger scale, these tend to be log–log plots that predict less well at smaller scales (see comments by Caraco *et al*., 2003 p. 349). So, as with Vollenweider's work in lakes, there are large-scale empirical relationships; nevertheless, they should not be used as guides to the restoration of individual rivers or lakes. Individual restoration trajectories do not often follow the ensemble average when nutrient loads are reduced or catchment land use is altered and time lags may be long (Phillips *et al*., 2005; Suding & Gross, 2006; Duarte *et al*., 2009).

What we have not yet been able to predict are the emergent properties of ecological dynamics and interactions in patchy and higher-frequency riverine systems with temporally evolving multivariate properties. Some argue that the complexity of the adaptive behaviour, the small scales of interaction and the dependence on initial conditions means that there are fundamental limits on our predictive abilities (Hauhs & Lange, 2008). Certainly, all the evidence points to nonlinearities, thresholds, and 'noisy' data (Allan, 2004) and to low power in the statistical relationships between land use measures and the riverine response (Murphy, 2010). Yet, plug flow and mass balance models of nutrient loads in rivers do work and can be reasonably accurate as long as the basic data contrive to capture the predominant (usually daily) fluctuations in concentrations and loads (Butturini, 2005). The ecological response of rivers, however, can be expected to be more complex relying, as it does, on the nonlinear, emergent, time-weighted response of the biota to the full spectrum of temperature, flow and loading variability. Also of importance is the fact that different components of the biota respond differently at different timescales (e.g. plankton, invertebrates, fish, macrophytes, and birds). Interestingly, Overton *et al*.'s (2010) meta-analysis of Australian river systems showed that some of the larger-scale, longer-term ecological components of floodplain river systems (birds, fish, macrophytes) are more predictable in their response to changes in environmental flows than are the smaller-scale, high-frequency-dominated components (algae, invertebrates). Also, Murphy (2010) found that the relationship between land use and river condition was much weaker if benthic invertebrates were used as indicators compared with the response of macrophytes. Thus, what is discernable at the sorts of scales that are normally sampled is very much scale dependent. It is necessary to resolve the full spectrum of the cross-scale advection–reaction–diffusion (A-R-D) dynamics – even out to inter-annual scales – especially if floodplain interactions and climate variability drive important interactions. This is the realm in which (hydrological) connectivity, and its modification by human agency, becomes important.

Characteristics of catchment loads

We do know that catchment exports change in predictable ways with aggregate changes in land use (Harris, 2002a,b, 2003; Caraco *et al.*, 2003; Smith *et al.*, 2003; Maybeck *et al.*, 2007). Oligotrophic waters fed by forested catchments have quite different dynamics than eutrophic waters fed by cleared, agricultural catchments (Harris, 1986, 2002a,b and Fig. 3). In the former, nutrient levels are low and inputs can be dominated by dissolved organic carbon and organic forms of N and P that are not biologically readily available. In the latter, readily available, inorganic nutrients predominate, and the relationship between land clearing and nutrient exports appears to be nonlinear with a threshold at about 50% reduction in the forested area in the catchment (Harris, 2002a, 2003). Beyond this threshold, catchment exports of biologically available nutrients increase rapidly and the dynamics of the receiving waters change accordingly from what Vollenweider called a 'low dynamic' state to a 'high dynamic' state (Harris, 1986). As far as we know, Vollenweider (1968) was the first to surmise that what he called changes in 'catchment physiology' were directly linked to ecological dynamics and outcomes in receiving waters.

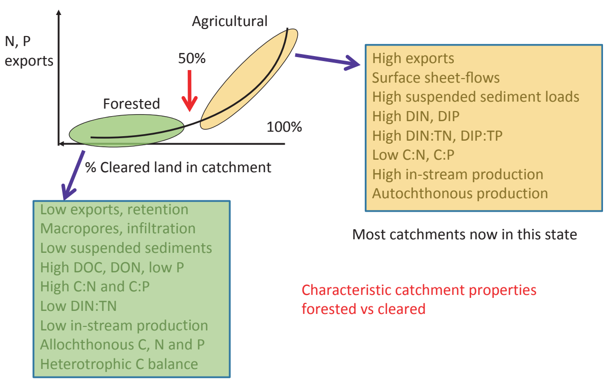

Fig. 3 Diagram of the changing nutrient export characteristics of forested and agricultural catchments. Full details and references can be found the study by Harris (2002a,b).

Agricultural catchments (or those in which agricultural land uses predominate) export more and different nutrient sources compared with forested catchments. We can only surmise that the flow paths also differ (Harris, 2002a, b) because clearing and ploughing change the distributions of macropores in soils, and drainage changes the spatial and temporal connectivity. The balance of surface flows and infiltration is changed so that the balance and forms of N and P (which tend to arrive by different pathways, Harris & Heathwaite, 2005) are also altered. Critically, once large nutrient surpluses have built up in agricultural soils and flow paths have been changed, it may be very hard to reverse the process of eutrophication; perhaps for decades or centuries (certainly timescales of recovery in rivers can be very long – see e.g. Burt *et al.*, 2010) It is now clear that many soil properties are both multi-fractally distributed in space and are self-organised (San Jose Martinez *et al.*, 2009). Even after revegetation, it may take many decades for critical soil properties to be re-established. Land clearing therefore destroys a number of complex and emergent properties of landscapes, none of which are resolved by the usual low-resolution sampling and mapping strategies employed by catchment managers and GIS-based modellers. The data we collected are not fit for purpose (Downes, 2010): they lack sufficient spatial and temporal resolution. Given that the ecology adaptively responds to the full spectrum of A-R-D reactions in both the catchment and the river system (hardly any of which is presently resolved), surprises are to be expected.

Characteristics of ecological models: making predictions under complexity

The arguments we have developed here so far have been designed to show that small-scale pattern and process (advection–reaction–diffusion) in catchments together with the complex adaptive response of the biota in both catchments (soils) and rivers lead to complex patterns in time and space, nonlinear, emergent outcomes and an unexpected level of informational or factual uncertainty.

Existing conceptual frameworks and models may assume the following (Lo & Mueller, 2010):

- Complete certainty: events can be described deterministically with deterministic forcing – risk and uncertainty are low and predictable outcomes may be expected. This is the world of Newtonian physics.
- Risk without uncertainty: events can be described deterministically with stochastic forcing – there is risk, but the probabilities are understood and may be dealt with statistically.
- Fully reducible uncertainty: there is uncertainty (there are things that we do not know), but the laws of large numbers can be used to apply statistical techniques to sufficiently large data sets. More effort and better data can be expected to provide solutions. This is the realm of the 'scientific method'.
- Partially reducible uncertainty: there is a limit to what we can deduce about the underlying generating functions from the data to hand. We might face (i) stochastic or time-varying parameters, (ii) nonlinearities too complex to be captured by existing data or techniques, (iii) non-stationarities and other features that render the Law of Large Numbers or Central Limit Theorems useless, causing statistical approaches to fail, (iv) dependence on unknown, multiple or unknowable drivers so that models suffer from parameter and structural errors (Harris, 2007; Beven, 2008a). This is the world of ecology and of socio-economic systems in which complexity, emergence and surprise are common (Lo & Mueller, 2010).
- Irreducible uncertainty: ignorance is rife. This is the extreme limiting case.

Here, we distinguish (following Knight, 1921) between risk (which is randomness that can be fully captured by probability and statistics and is therefore reducible) and Knightian uncertainty (all other forms of randomness) which – as defined previously – is different in degree and represents ignorance about significant processes and parameters (see also Beven, 2008a). We have presented evidence here that aquatic ecosystems – and, of course, the socio-economic systems in which they are embedded – fall into category four (the 4th bullet point above). In complex systems, informational uncertainty is only partially reducible.

The arguments in this paper all point to the same problem as that which contributed to the Great Financial Crash (Lo & Mueller, 2010); the assumption that a world in which uncertainty is only partially reducible can be represented by concepts and models (and managed by practices) in which uncertainty is fully reduced!

In practice, which category the concepts and models fall into actually depends on the scale (time and space) of the model and the scale of pattern and process in the real world (and, of course, whether the data are able to adequately describe what is going on). Much depends on the ability to separate scales (Fig. 4). For example, Kirchner (2009) was able to inverse a model catchment (by effectively 'doing hydrology backwards') only by making a number of strong assumptions about constancy of catchment characteristics during the period of interest. If it is possible to categorise large-scale processes as constants or slow deterministic drivers, middle-scale processes by deterministic relationship in a (conceptual) model and small-scale processes by empirical or

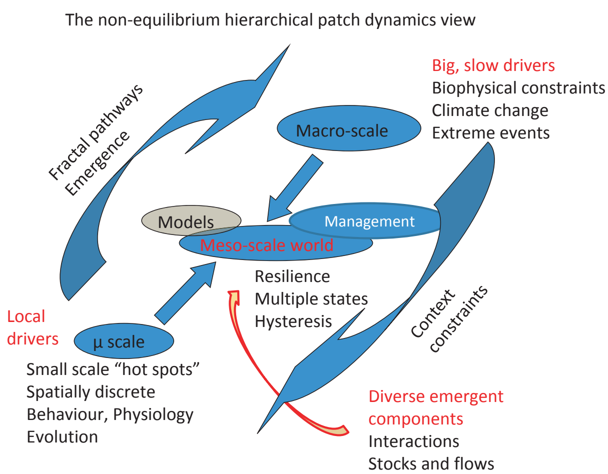

Fig. 4 The world of environmental management is the meso-scale world of human scale and of experiments and modelling. This is sandwiched between the non-stationary global-scale drivers (including biogeography and evolved stoichiometry) and the climatological scale on the one hand and the micro-scale world of spatially discrete (and fractally distributed) pattern and process on the other. In a world of complexity of feedbacks and emergence, there is both upward and downward causation (Wimsatt, 2007). The meso-scale world shows resilience and multiple states in response to these cross-scale drivers.

other parameterisations – and if it is the middle-scale dynamics that are the ones for which predictions are required, then some progress can be made. If the small-scale ecology and biogeochemistry of catchments and rivers are truly complex, then uncertainties must remain and aspects of it will not be computable (Carpenter *et al.*, 2009). We should expect surprises. Small-scale A-R-D dynamics coupled with the diversity and biology of aquatic ecosystems ought to show complex responses to multiple stressors because of the nonlinear, adaptive and time-dependent responses to the spectrum of environmental variability, as well as trophic and other cross-scale system-level interactions (Fig. 4). It really is important to know how much of the catchment restoration problem is truly complex (and therefore not computable even with better knowledge) rather than being merely complicated (and therefore computable if we had better knowledge of the components and drivers). The strong suspicion remains that it is the former.

The meso-scale world of the biosphere and the anthroposphere is normally messy and complex with many adaptive, cross-scale interactions (Wimsatt, 2007). Confusion and lack of clarity occur where data are sparse, where species abundances are of interest, where nonlinearities occur, where there is strong dependence on initial conditions (founder effects and temporally evolving properties) and where scales overlap such that the effects of multiple stressors cannot be separated: this is the usual situation in rivers. The question remains the same as that raised by Harris & Heathwaite (2005): To what extent does the small-scale patchiness caused by A-R-D relationships in catchments and rivers lead to complexity and emergence in the biology, and therefore surprises?

The key conclusion that we draw from the above analysis is ecological systems are constrained, complex and emergent non-equilibrium systems driven by climate and other drivers from scales of days and weeks out to evolutionary scales. Change is constant: everything is on a trajectory to somewhere (Botkin, 1990; Duarte *et al.*, 2009). There is much fundamental epistemic (Knightian) uncertainty and lack of predictability ability across all scales (Beven, 2008a). All models will be wrong – some will be more wrong than others: predictions will be flawed. We cannot substitute model predictions for a lack of data. Given the nature of the adapted, evolving ecological entities involved, we should not expect Environmental Directives to work well. If river restoration involves systems of coupled ecological, socio-economic actors, then any presupposed 'predict-act' framework should be fundamentally flawed. To quote Beven (2008a), it really is an 'uncertain future'.

This is an area of science and management that is quite fraught with controversy. For example, Keith Beven, a key proponent of the analysis of uncertainty in catchment and hydrological models, has been accused in print of 'undermining the science' (Beven, 2006, 2008b). There really are strong framing assumptions in this area; the inbuilt assumptions in modelling and design described by Giampietro *et al.* (2006) and the 'group think' of which Carpenter *et al.* (2009) speak do determine the design and analysis of models and restoration experiments. Unless we can acknowledge the underlying uncertainties and come to grips with the analysis and management of investment under uncertainty, little progress will be made. If we can do this, then the effectiveness of programmes of measures will increase and restoration programmes will achieve greater success. Perhaps, we may begin to arrest the global decline in freshwater biodiversity.

Managing (and investing) under uncertainty

So now we turn to the problem of coupling informational uncertainty to the right-hand side of Fig. 1 – the normative uncertainty – the state of doubt as to what to do (Newig *et al.*, 2005). Figure 5 diagrammatically displays the spectrum of possibilities from complete certainty to complete uncertainty and places ecosystems and environmental management in this context. Ecology and ecological restoration occupy a fundamentally uncertain space on the right of the diagram. Even (the much vaunted) adaptive management requires repeatable responses to management actions for it to work, so that there is feedback between action and outcome. Clearly, and for a number of reasons, there are many occasions at present when there is no feedback from the ecological state after management action. Ecology is not physics: all models and measurements are an abstraction from a small- to meso-scale complex entity (Fig. 4, see also Holdgate & Beament, 1975). Ecologists must not fall into the realist trap of assuming that their data and models represent reality – important aspects of ecological complexity are not captured by temporally and spatially sparse data or by present models.

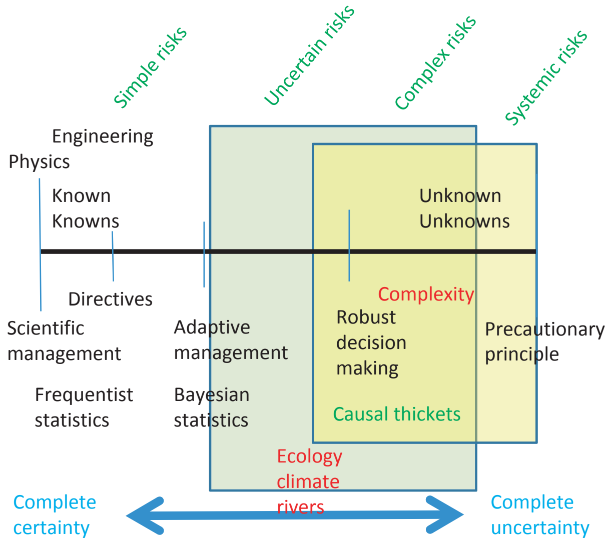

Fig. 5 There is a spectrum of uncertainties in nature from certainty (and the security of 'predict-act' prescriptions) to complete uncertainty (and the necessity of precautionary principles). The use of management frameworks like Environmental Directives assumes a large number of 'known knowns' and low epistemic (Knightian) uncertainty. As uncertainty increases, the adaptive management and Bayesian techniques are possible where there are predictable responses and where priors can be defined. The space of complexity (where global priors are undefined) moves towards Rumsfeld's 'unknown unknowns', Taleb's 'Black Swans' (Taleb, 2007) and the need for Robust Decision Making (Lempert *et al.*, 2010).

In the absence of models able to predict 'everything everywhere', what can be done? How do we obtain better evidence so that we can invest under uncertainty? Reframing the question in a more complex light can help us to start down a more realistic path. Once we recognise the importance of small-scale pattern and process in these systems and the cross-scale, non-stationarity of the ecological response, then we can begin to design more effective monitoring programmes.

First, we need better evidence of even incremental change in ecosystem responses. Some of this more partial evidence might come from high-frequency monitoring and a better understanding of the true scales of pattern and process in catchments and rivers. We can use new technologies to exploit the 'high-frequency wave of the future' (Kirchner *et al.*, 2004). New results are, at least, giving a clearer view of event-scale impacts (Jordan *et al.*, 2007). If high-frequency monitoring data can give information about the approach to tipping points, then it might be possible to begin to unpick the causal thickets that we frequently encounter.

New data are beginning to show that small-scale connectivity in catchments and rivers has a strong impact on the biota. Processes and interactions at scales of tens to hundreds of metres – connectivity between critical source areas strongly linked to the channel – produce identifiable ecological impacts (Lane, Reaney & Heathwaite, 2009; Peterson *et al.*, 2010; Reaney *et al.*, 2010). Thus, we are beginning to scale down to the true scales of connectivity – well below our usual scales of monitoring – to begin to resolve which parts of catchments impact most on the ecology of rivers. If we can develop this work, then perhaps we can reduce uncertainty and reconceptualise risk in terms of connectivity at relevant ecological scales by focussing on what matters where.

Second, we should seek new measures of ecological responses based on the properties of complex ecosystems. 'We need to explore further the use of integrative measures of river health, and focus on establishing a link between the measure and impaired ecological integrity. Ecosystem level variables… show promise' (Boulton, 1999). Rather than monitoring well-known population and community statistics and other 'noisy' symptoms, we should seek system-level indicators of changing ecological process and integrity. For example, Bunn, Davies & Mosich (1999) employed a number of bulk ecosystem measures of respiration and photosynthesis as measures of system health. We may need to go as far as defining what might be called ecosystem 'order parameters' or invariants that describe and allow predictions to be made about the future system state (Ramos, Altshuler & Maloy, 2009). This requires a major rethink of ecosystem structure and function in the context of complex systems theory.

Similarly, there is scope for what might be called 'complexity indicators' (Muller, 2005; Parrott, 2010), new indicators designed to resolve the properties of the complexity with which we must deal. Much of this work comes, at present, from the area of theoretical computer science, so it is not usually transparent to ecologists; nevertheless, developments in such areas as fluctuation complexity and minimum information gain show promise for applications in ecology and catchment science (Hauhs & Lange, 2008).

Third, we need to look for indicators of ecological progress that could be used to indicate even some movement in the right direction. It would be better, perhaps, to monitor measures of partial success (on a recovery trajectory of some kind) rather than (as now) 'go for broke' on the ultimate ecological outcomes that will be difficult to observe and may be a long time coming. Adaptive management requires a link between action and outcome; at present, low power provides little guidance. Here, we need to rethink our objectives, indicators and values and do more 'joined-up thinking' when we design policies and incentives to restore land use and river condition. By this, we mean designing programmes of measures so as to achieve multiple system-level outcomes from individual measures and also making sure that from the outset we get a realistic understanding of pattern and process linked to the programme design and implementation, together with a realistic set of desired outcomes. At present, we have too much fragmentation of indicators, monitoring programmes, measures and policies: a set of indicators and measures that take ecologically relevant scales and complex interactions into effect has a better chance of success. We urgently need to reassess our framing assumptions.

So we need to find evidence of some response to programmes of measures, even if we cannot shift the system back to 'good' ecological condition as required by, for example, the EU Water Framework Directive. Such evidence might come from a closer examination of what is now regarded as 'noise' in ecological data. Such 'noise' is actually evidence of complex dynamics and, rather than averaging it out as now, we should treat it as evidence of the resultant of small-scale adaptive interactions (Harris & Heathwaite, 2005) – effectively the result of many time-dependent evolving biological properties (Hauhs & Lange, 2008). Rather than destroying information by averaging (Harris, 2007), a complex frame of reference sees such data as resulting from non-equilibrium 'sum over paths' with observable statistical properties.

Fourth, we should acknowledge the pervasive uncertainty and act accordingly. The tools of Robust Decision Making can be exploited to provide a guide to action, even under great uncertainty (Lempert & Collins, 2007; Lempert, Popper & Bankes, 2010). Thus, the development of 'no regrets' actions that might move the system in the right direction, even when a response is either very uncertain or not even expected, is a step in the right direction. Allen, Tainter & Hoekstra (1999) have suggested that perhaps the only way to manage complex, adaptive and nonlinear systems is to manage the context, i.e. manage the bigger picture constraints on the system and acknowledge the uncertainties in the actual response. To this extent, 'win-win' catchment management policies are a step in the right direction, and river basin plans can certainly be regarded as no regrets polices of context management. Without better evidence of (even partial) ecological improvements then, such initiatives will be resisted by catchment land holders unless 'win–win' measures can be instituted.

Finally, perhaps, we should acknowledge that in agricultural catchments where soil nutrient levels are typically high, and further inputs are expected from both land-based and atmospheric sources of, for example, reactive nitrogen (and the catchment is already in a 'high dynamic' high export state, Fig. 3), given a growing human population and the requirement for increased food production, it is unrealistic to expect much ecological improvement in the condition of rivers. Heroic changes in land use and management practices (a return to more that 50% perennial vegetation, Fig. 3) are probably out of the question in most countries in the 21st century. 'Natural' may not be the appropriate goal for water management in this century (Bishop *et al.*, 2009).

Above all, we require a means to prioritise investments in an uncertain world. As we move into an era of global and climate change, risks are becoming ever more complex and systemic (Deere-Birkbeck, 2009). Beven (2007) and Beven & Alcock (2011) has suggested the use of partial 'models of everything everywhere' (known to be wrong in many aspects) not as prediction engines but as a rejectionist framework to test hypotheses and to investigate the 'particularities of place'. In the absence of suitable landscape-scale experiments (although see Lane *et al.*, 2009; Reaney *et al.*, 2010), we can at least try out our best guesses and quantify the uncertainties. Above all, it is time to acknowledge the underlying uncertainty and complexity of an evolved and adaptive natural world and to work with it to achieve the best results we can. Denial of complexity (Wynne, 2005) gets us nowhere.

Ultimately, what is required is the acceptance and understanding that this is truly a post-normal science (Ravetz & Funtowicz, 1999; Harris, 2007) where culture, beliefs and values are features of both the informational and normative uncertainty. The ability to manage and invest under uncertainty requires better analytical tools as well as improved participatory tools together with the acceptance of non-computable behaviour in catchments and aquatic ecosystems. Predictive power will remain poor so that decision-making is constrained to the worlds of adaptive management using complex indicators and of robust decision making and 'no regrets' policies.

Better evidence at appropriate scales, progress indicators, deliberative processes, coproduction of knowledge and an acknowledgement of uncertainty coupled with 'no regrets policies' may help us to generate more 'power' in the relationships between catchment management and ecological outcomes in rivers; but more data will only raise more questions, perhaps at finer scales. Above all, we must recognise and work with the irreducible uncertainties; 'more data and better models' will not suffice to completely reduce the investment risks, and we will always have to live with both informational and normative uncertainty. This is not a 'physical' problem but one involving living, adaptive beings (Hauhs & Lange, 2008).

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