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Uncertain restoration of gravel-bed rivers and the role of geomorphology

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Abstract

River restoration projects in gravel-bed rivers are becoming increasingly sophisticated and complex as river managers and scientists attempt to deliver the goals of catchment-scale ecosystem restoration. With increased sophistication, come the dual challenges of recognizing and responding to the uncertainty inherent in the restoration process. Uncertainty is rarely explicitly recognised in current restoration projects and, where it is, the scope and definition are limited. In this paper we argue that uncertainty is a fundamental element of river restoration and that the sources of uncertainty are varied. A typology for understanding and communicating uncertainty in terms of these sources is presented. One of the myths surrounding uncertainty is the notion that being uncertain is the same as not knowing anything. In fact, when uncertainty is expressed as a statement of plausible outcome and/or significance, expressing uncertainty is a very informative statement of knowledge. The significance of uncertainty is explored conceptually and quantified for two contrasting examples from two gravel-bed river restoration projects. Respectively, these demonstrate that uncertainty in the conceptual model applied to a restoration project can have significant impacts on the restoration process and that unreliability uncertainties can affect the design of bankfull channel dimensions. The paper concludes with a discussion of the approaches to incorporating uncertainty in river restoration projects, and argues for one that embraces uncertainty. We present an approach for embracing geomorphic uncertainty in physical habitat restoration, that uses coupled habitat and landscape evolution models to define the plausible outcomes for a given restoration project.

1. Introduction

River management philosophy over the past decade has converged on the over-arching ethos of sustainable management of water and associated ecosystem functions within the spatial context of the river catchment (Graf, 2001; Ward et al., 2002; Downs and

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Gregory, 2004). Embedded within these ethos are the traditional river management aims of protection against damaging floods and the provision of adequate water supply to the human populations within the catchment (Fleming, 2002). However a range of "new" concerns have also emerged that relate explicitly to the functioning of aquatic ecosystems. These include preservation of physical integrity (Graf, 2001; Everard, 2004), restoration of ecosystem functions (Richards et al., 2002) and management of water, nutrient and sediment fluxes at the catchment scale. As a result, approaches to river management are now more holistic in their conceptualisation (Everard and Powell, 2002; Newson, 2002), multidisciplinary (Fleming, 2002) and participatory in their implementation (Clark, 2002).

River restoration is a key component of this new river management, widely seen as the process through which river basin management's aspirations and targets will be delivered (NRC, 1999; European Council, 2000). Thus, a tremendous diversity of river restoration projects have been undertaken throughout the world (Boon et al., 1992; Calow and Petts, 1994; Brookes and Shields, 1996; Iversen et al., 1998; Malakoff, 2004) in response to a well-documented array of adverse impacts from anthropogenic disturbances (Brookes, 1988; Coltorti, 1997; Graf, 2001; Knox, 2001). The restoration science community has responded to the large demands (from the practitioner, policy-maker and stakeholder communities) for ways to restore and mitigate such problems with a rich assortment of approaches, strategies and tools (e.g., Sear, 1994; Brookes, 1995; Brookes and Sear, 1996; FISRWG, 1998; Wissmar and Beschta, 1998; Gilvear, 1999; Koehn et al., 2001).

As restoration evolves, projects are becoming more expensive, complex and technically difficult, with lifetimes now extending over geomorphologically relevant timescales (Newson, 2002; Sear and Arnell, 2006). With increasing sophistication comes additional risks in terms of setting and meeting realistic project targets and to communicate complex models of river environments to stakeholders. Indeed, the results from recent monitoring programmes are beginning to cast doubt on the ability of restoration projects, as currently practised, to deliver some of these targets (Harrison et al., 2004; Williams et al., 2004).

Central to progressing more sophisticated models of river restoration is our ability to comprehend and communicate the uncertainty in the science to a stakeholder base (that may include other scientific disciplines) that has been brought up with the notion that environments can be managed in a deterministic fashion. Uncertainty exists throughout the restoration process (Fig. 28.1a), yet paradoxically, most restoration projects fail to explicitly identify or communicate the uncertainty (Wheaton et al., 2006). The scientific community implicitly accepts, and to a certain extent thrives on, the inherent uncertainties in conceptual ideas, approaches, tools and strategies it provides (Pollack, 2003). However, restoration science has largely failed to transparently communicate these uncertainties to restoration practitioners, policy-makers and stakeholders (Wissmar and Bisson, 2003).

Presumably, either adaptive management (encouraging the treatment of restoration projects as 'learning by doing') or a lack of long-term monitoring have prevented the emergence of any major consequences from ignoring uncertainties (Walters, 1997; Clark, 2002). Should long-term monitoring actually take place and reveal a systematic pattern of project failure (e.g., Kondolf, 1995; Kondolf et al., 2001; Downs and

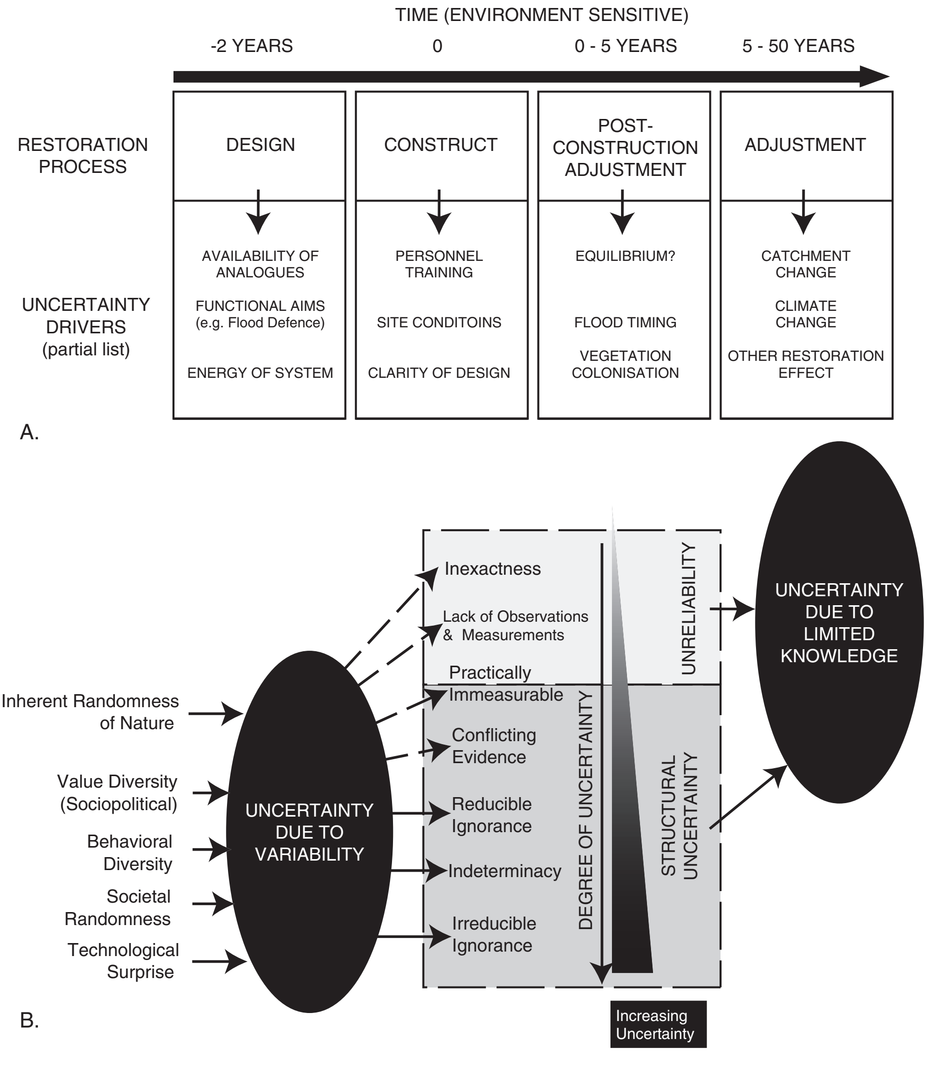

Figure 28.1. Uncertainty conceptualised. (A) Sources of uncertainty in the restoration process. (B) Typology for uncertainty based on sources of uncertainty. (Adapted from Rotmans and Van Asselt, 2001.)

Kondolf, 2002), stakeholders and policy-makers may well demand explanations from restoration scientists and practitioners (Wilcock, 1997). If such a process also reveals the plethora of uncertainties in restoration objectives, strategies and approaches that have to date been ignored, then the community might well be accused of gross negligence. Or, more to the point, the restoration community may suffer funding cuts or be sued.

To address these issues, we herein wish to provide a context for understanding uncertainty in gravel-bed river restoration. Our objectives in this paper are twofold. First, we shall demonstrate that a variety of sources of uncertainty exist in river restoration and that their significance is context specific (see Johnson and Rinaldi, 1998). Second, we shall contrast a variety of approaches to deal with uncertainty. From the outset, it is important to distinguish that our primary concerns about uncertainties as they relate to river restoration are not about the uncertainties themselves; but rather the implications of the failure of the restoration community to communicate them. The basic premise is that this leads to unrealistic expectations among stakeholders about the outcomes of restoration projects.

2. The concept of uncertainty

Although uncertainty is a fundamental and basic concept in the sciences (Jamieson, 1996; Pollack, 2003), we are often careless with its semantics. This carelessness makes it easy to confuse uncertainty with a complete lack of knowledge. For our purposes, we will consider a broad definition of uncertainty proposed by Van Asselt (2000), in which uncertainty is defined in terms of its source. Van Asselt (2000) considers uncertainty to stem from two sources – limited knowledge (as opposed to a lack of knowledge) and variability (Fig. 28.1b). Uncertainty due to variability includes natural variation, natural variability (spatial and temporal), non-linear, random and chaotic behaviour, as well as value diversity within a society (Rotmans and Van Asselt, 2001). Paradoxically, *uncertainties due to variability* contribute to *uncertainty due to limited knowledge*, which can be segregated into *structural* and *unreliability uncertainties. Unreliability uncertainties* are those most typically quantified by scientists (e.g., error in a measurement). However, *structural uncertainties* are typically of a higher degree than *unreliability uncertainties* and represent more fundamental barriers to understanding (Van Asselt and Rotmans, 2002). *Conflicting evidence* is one type of *structural uncertainty*, which is particularly prevalent in river restoration for deciding which approaches or what types of analogues to use. There exist also *uncertainties due to reducible and irreducible ignorance*, which represent the difference between things 'we could but don't know', and things 'we cannot know' (Van Asselt and Rotmans, 2002). We will use this typology of uncertainty throughout the rest of this paper to identify the specific uncertainties discussed, and a description of the terms is provided in Table 28.1 to assist the reader. Where a specific source of uncertainty is classified according to the typology, we will use *italics*. Additionally, where we are generically interested in future outcomes (i.e., prediction) we will call the set of all plausible outcomes the 'plausible outcome space.'

3. Significance of uncertainty

As pointed out in the introduction, uncertainties are ubiquitous in river restoration. If one assumes that uncertainty is unequivocally a negative attribute, this paints a very bleak outlook for river restoration. However, under the broader definitions of uncertainty described in the previous section (e.g., Van Asselt and Rotmans, 2002), to identify our uncertainty about something is actually a statement of our knowledge.

Table 28.1. Glossary of terms used in figure.

Uncertainty typology	Term	Description
Structural uncertainty	Irreducible ignorance	*Things we cannot know*
	Indeterminacy	*Things we will never know*
	Reducible ignorance	*We do not know what we do not know* – this is information/knowledge that is accessible but as yet we have not as a community discovered it.
	Conflicting evidence	*We do not know what we know* – knowledge is inexact and different interpretations of the same phenomena/information exist.
Unreliability uncertainty	Practically immeasurable	*We know what we do not know* – refers to phenomena or measurements that cannot as yet be undertaken perhaps because of scaling issues or the technology available at the time.
	Lack of observations and measurements	*Could have, should have, would have…but did not*. A real aspect in most projects is the pieces of information that we were unable to record or simply did not record.
	Inexactness	*Related to error, imprecision and accuracy* of the information/ measurements acquired.

Source: Adapted from text of Van Asselt and Rotmans (2002).

A pedagogic exploration of the semantics of uncertainty without considering the significance of uncertainty is of limited applied value. The significance of our uncertainty depends entirely on the context and perspective from which we consider it. For example uncertainty in nutrient loading may be an eminent concern in a system being restored for water-quality purposes; yet may be of little consequence in the same system being restored for flood protection purposes. Where restoration is driven by a narrow set of stakeholder, agency or societal objectives (e.g., single species salmonid restoration) the significance of uncertainty is relatively simplified. Increasingly, however, restoration practitioners have been considering more sophisticated, multipurpose objectives simultaneously (catchment restoration, ecosystem restoration, dynamic system restoration, etc.)

With more sophisticated restoration objectives come additional sources of uncertainty with interdependent significance. Returning to the example above, if the system were being restored for both flood protection and water-quality purposes, all the sources of uncertainty deemed to be significant for either objective would have to be considered. If one were trying to communicate their uncertainty about nutrient loading to a group of water-quality scientists, reporting significance in terms of nutrient concentrations would be perfectly acceptable. However, if one were communicating the same uncertainty to a lay audience of stakeholders at the local fishing club, a more effective means of communicating the significance of that uncertainty might be in terms of the potential impacts of uncertain nutrient loading on fish.

It is premature to say too much about whether uncertainty is significant in restoration because there has been very little research on the topic (Wheaton, 2004; Wheaton et al., in press). Basic reasoning is adequate to identify the ubiquity of uncertainty in restoration. However, until the significance of those uncertainties is considered, it is difficult to say whether it matters or not. Unfortunately for those looking to make generalizations, significance is a value-laden and context-specific consideration. Although this makes significance a difficult characteristic to generalise about, it also provides interpretations of uncertainty that can have greater practical utility. Hence, here we can only provide the reader with a couple of specific case studies of the significance of uncertainty in river restoration. We can, however, draw a precautionary conclusion that based on the ramifications of the uncertainties we know to exist, the plausible outcome space of restoration activities certainty includes outcomes that could do more harm than good. Regrettably, the restoration community has instead chosen to assume that because its intentions are 'good', the outcomes cannot be worse than the status-quo. This should not be misinterpreted as a general scepticism about restoration. Rather, as we show in the discussion, it means that restoration science should embrace uncertainty in order to help the restoration community and general public form more realistic expectations about restoration. First, we will explore these concepts with two specific examples common to many restoration projects: (a) conceptual models and (b) reference conditions.

3.1. Knowledge uncertainty and the role of the conceptual model in river restoration

Conceptual models of how river systems function lie at the heart of restoration projects. They are used to inform the understanding of river system function and river system form. In turn, these affect the design and management of the restoration project. Wheaton et al. (2004) argued that numerous conceptual models exist within the scientific literature that might be selected for a given restoration project; but, they advocated using these only to develop multiple working design hypotheses that could be tested and refined prior to constructing a restoration project based on them. Rutherfurd et al. (in press) concluded that however powerful the conceptual model may be, it brings with it high risks if it has poor validity or is just plain wrong with respect to the given river system. This notion is not unfamiliar; for example Lewin et al. (1998) argued against the validity of applying regime models to gravel-bed rivers in an era of environmental change, preferring instead to take a more long-term view of river channel dynamics. In the van Asselt (2000) typology, Lewin et al.'s (1998) argument for extending the temporal scale of investigation of river management projects was intended to reduce *uncertainty due to natural variability* by quantifying the nature and drivers of natural variability. More recently, conceptual models that highlight the importance of natural variability have been embraced as an essential feature of river restoration projects (Wissmar and Bisson, 2003; Hughes et al., 2005). The example below illustrates some of the uncertainties and pitfalls of using conceptual models in river restoration.

The restoration of salmonid spawning habitat in the UK rivers is driven by a well-documented decline in stocks (WWF, 2001). Despite widespread evidence for the importance of marine factors, river management agencies have applied a conceptual model based on the premise that recruitment is constrained by the quality of the incubation environment within the freshwater phase of the salmon life cycle (Reiser, 1998). This is not surprising when you consider that river management agencies have little or no jurisdiction in the marine environment. The conceptual model correlates fine sedimentation of spawning gravels directly with egg survival (Fig. 28.2a). Management response has been driven by this conceptual model to undertake investment in gravel cleaning, channel narrowing, bank erosion control and catchment-scale treatment of soil erosion, (Greig et al., 2005). Furthermore, considerable investment is made in assessing spawning habitat quality based on indicators that measure the proportions of fine inorganic sediment within the gravels (Reiser, 1998). In the UK, a general absence of monitoring has led to the widespread assumption that these approaches to river restoration are adequate (McMellin et al., 2002) and meet the implicit restoration targets of increasing survival rates to swim up. Thus, a lack of monitoring means it is unknown whether salmon stocks are improving, declining further, or not responding at all to restoration efforts. This is an example of *uncertainty due to reducible ignorance* (refer to Fig. 28.1b), whose significance arguably undermines both the effectiveness of past restoration and appropriateness of future restoration.

From a scientific perspective, the recruitment-constrained conceptual model has been challenged by new research on the factors affecting the incubation of salmonids within UK river gravels (Greig et al., 2005). In a study of four contrasting river environments, Greig et al. (2005) documented widespread variability in salmon egg survival. These were attributed to a suite of factors, including sedimentation by inorganic material, oxygen depletion due to oxidation of organic sediments, reduced intragravel flow rates due to low flows, and potentially, decreased oxygen concentrations due to upwelling of groundwater with low dissolved oxygen content. The research also tested the effectiveness of a range of popular sediment-based indicators of incubation habitat quality utilised by fisheries managers. The popular indicators were demonstrated to be ineffective since they failed to represent the complex factors leading to poor survival (Greig et al., 2005).

The development of a new conceptual model from Greig et al.'s (2005) research highlights the multiple and interacting factors that can influence the survival of incubating salmonids (Fig. 28.2b). The improved scientific understanding arising from this model results not in a reduction in *uncertainty due to limited knowledge*, but rather a transformation from *uncertainty due to reducible ignorance* to uncertainties of lesser significance (see Fig. 28.1b) and of more information value (Van Asselt and Rotmans, 2002):

- *uncertainties due to unreliability* – current restoration projects do not measure the necessary variables (but we now know what they are) and the variables when measured are prone to measurement errors
- *structural uncertainties due to conflicting evidence* – empirical observations within the scientific literature strongly support the importance of inorganic sedimentation on salmonid egg survival (Reiser, 1998)

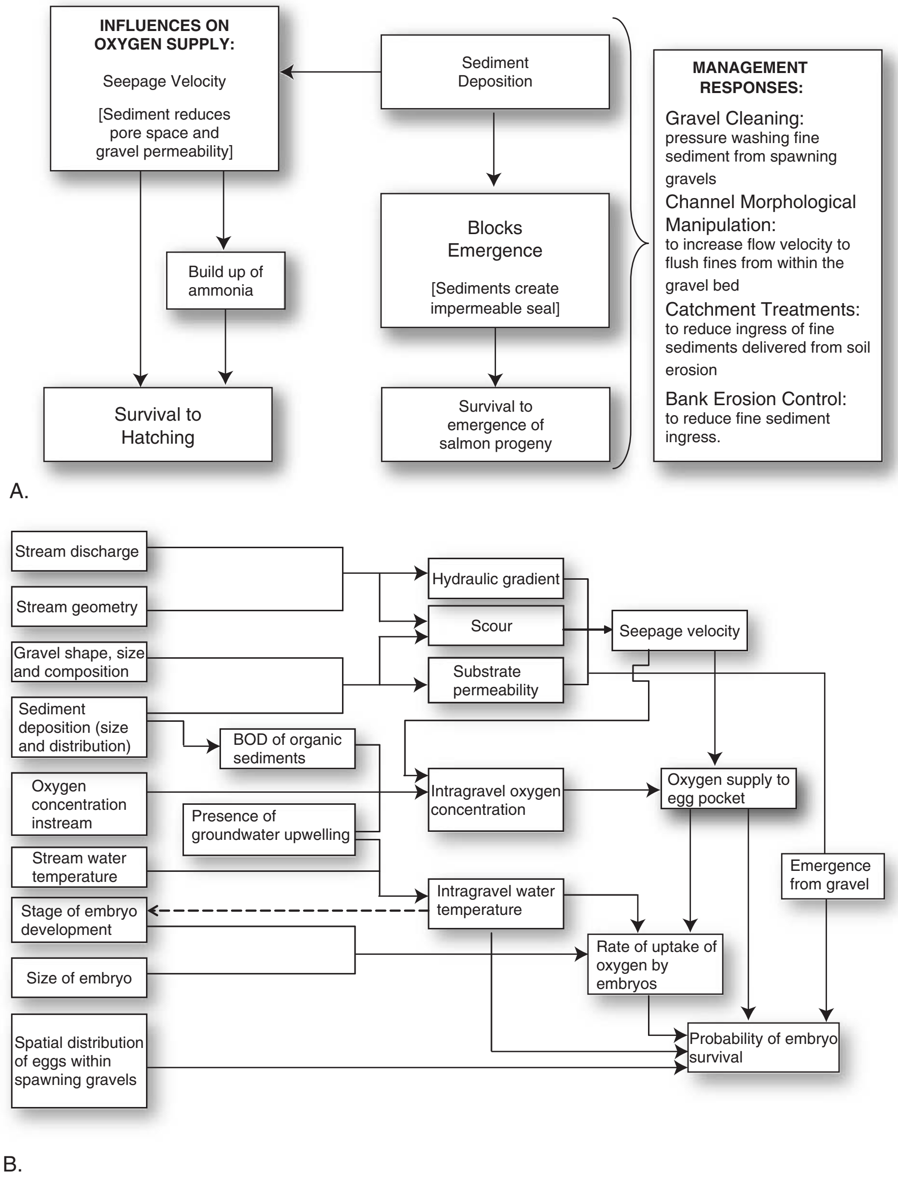

Figure 28.2. Contrasting conceptual models of spawning incubation. (A) The conceptual model underpinning a suite of current restoration practices. (B) A conceptual model of spawning incubation that results in different potential causes than the currently adopted model.

- *uncertainty due to variability* – inherent randomness of nature exhibited in the dynamics of physical and chemical processes

This so-called 'transformation' of uncertainty can be viewed as an increase in uncertainty, as further study only revealed further uncertainties. However, in terms of the significance of these uncertainties, the additional uncertainties actually focus attention on the most important factors of the incubation problem and act to provide helpful information. The original conceptual model (Fig. 28.2a) was introducing a *structural uncertainty due to reducible ignorance* as it was being applied. In other words, it was not known whether the model was correct, but in principle it could be known by testing the model. The testing of the conceptual model and refinement of a new conceptual model is not free of uncertainty, but confines this limited knowledge to a much narrower plausible outcome space.

3.2. Unreliability uncertainty and the design of reference condition channels

River restoration often requires some degree of channel design. The significance of uncertainty in the design process is seldom quantified, and attitudes to uncertainty vary depending on the level of accuracy demanded by the project. Many restoration projects (e.g., those that involve significant changes in flood risk as is the case when a channel is reconnected to its floodplain) might require considerable design accuracy to ensure an outcome that satisfies stakeholders. We contend that there is a systematic distinction in the required design precision between upland and lowland gravel-bed river environments. In high-energy upland systems, rates of morphological adjustment are frequently sufficient that designs based on 'passive' approaches represent cost-effective means of restoring or rehabilitating the functional characteristics of substantial lengths of river channel (Brookes and Sear, 1996). In comparison, lowland gravel-bed rivers systems often have a combination of low stream power and resistant channel boundaries that do not allow natural recovery in anything less than unacceptably long timescales (Brookes and Sear, 1996). In such cases the legacy of channel design mistakes would, without corrective intervention, last for many years, implying that a high accuracy in the restoration design is required, as well as in the implementation of that design. Moreover, lowland systems are frequently heavily disturbed and are often those systems most in need of restoration (Sear et al., 2000).

A restoration scenario typical of lowland river restoration in the UK was investigated to quantify the magnitude of uncertainty in the final design arising from *unreliability uncertainties due to inexactness and lack of observations and measurement*. The river Cherwell is a low gradient, gravel-bed river confined within cohesive floodplain alluvium. The Cherwell has a long history of modification to its hydrological regime, surrounding catchment land cover and management, planform, cross-section and long profile (for the complete site description and study, see Sear et al., 2001). The restoration design for the Cherwell was theoretical, and formed part of a hydrological modelling study to determine the effectiveness of different channel and floodplain restoration scenarios on downstream flood peaks. An element of this wider project necessitated the reconstruction of the channel dimensions and floodplain vegetation structure prior to major channel modifications and land management in the 19th century. The initial approach was to identify semi-natural reference sites from which an empirical hydraulic geometry model might be developed. This method itself is prone to *structural uncertainties* that question its validity. Assuming that reliable data are available (*unreliability uncertainties*), power functions relating channel geometry to drainage area can therefore be developed and used to predict values for any given point in the river network. In the case of the river Cherwell, values of discharge and depth for the pre-disturbed watercourse were not available due to the absence of suitable analogue reaches and historical data sources. Paradoxically, this condition is typical of heavily disturbed lowland gravel-bed rivers. As a way around this, we estimated the pre-disturbance channel dimensions with a combination of empirical methods and analytical modelling techniques. Given the methodological uncertainties in the work-around method, we explored their significance in terms of the designed channel dimensions.

3.2.1. Estimating pre-disturbance bankfull channel width

Values of channel width were derived with a relatively small degree of *inaccuracy uncertainties* from large-scale historic maps (Downward, 1995), with catchment area used as a surrogate for discharge (Sear, 1996). Downward (1995) defines two main categories of error in deriving map-based estimates of channel boundaries: inherent errors and operational errors. Inherent errors are included at each point in the process of data capture from landscape to final visualisation on the map (*unreliability uncertainties*), including those arising from the surveying process itself. Downward (1995) provides a method for estimating the component of inherit error arising from the process of registering the paper map into digital format (*unreliability uncertainties*); but, since we were not comparing locations of channel boundaries at different points in time, positional accuracy were not relevant. It was assumed that inherent error, the result of survey and cartographic errors at the time of map production (and the source of *structural uncertainty* in the method) was largely unquantifiable.

Unreliability uncertainties in the form of operational errors occur in the estimation of channel width from the raster map. These arise from the pixel representation of the channel boundary. For this study, the pixel area was set to $1\,m^2$ since this corresponded to the approximate thickness of the lines denoting channel boundaries on the original map. Representation of the channel boundary usually involved only one pixel, but where channel curvature was significant, this increased to two pixels. The centre of each pixel, or the boundary between two pixels, was taken as the location of the channel boundary. Using this standard approach, the channel width could therefore be said to lie within ± 1.0 m for locations where the channel boundary was represented by a single pixel, or ± 2.0 m for locations where the line was represented by two pixels. Quantifiable total mapping error (TME) was therefore estimated to be ± 1.0 to 2.0 m. This value represents the summation of all recognised *unreliability uncertainties* in the method. To provide estimates of bankfull width for the pre-engineered channel, nine sites were identified that had a natural planform. At these sites, a length of channel of up to 200 m was defined, and 15 channel widths were digitally measured using the on-screen distance tool. The channel widths were divided into bends and inflexion points. The error bands shown on Fig. 28.3a are, however, substantial with average errors of ± 1.3–2.2 m.

3.2.2. Estimating pre-disturbance bankfull channel depth

Having defined pre-engineered width values, channel depth was estimated using a rational regime type model, VARSLOPE (Millar and Quick, 1993, 1998).

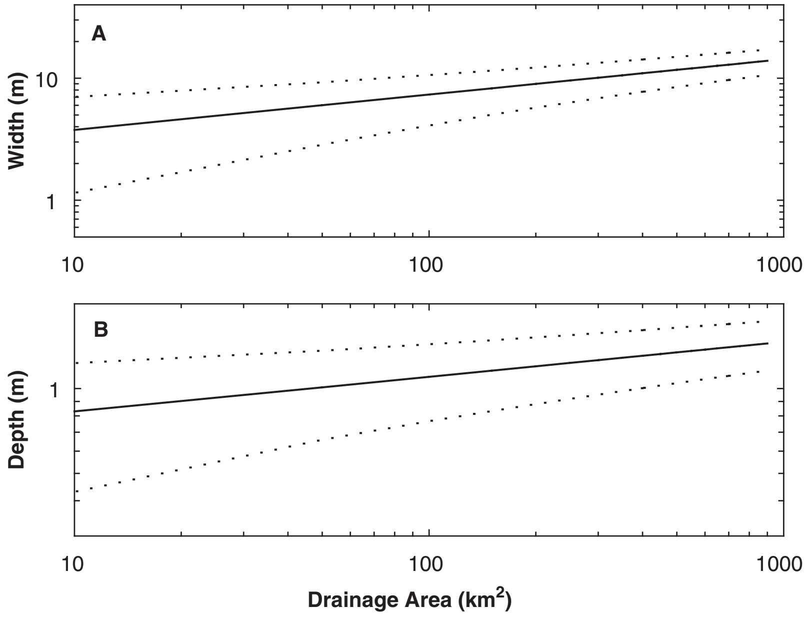

Figure 28.3. Uncertainty bounds around estimations of channel bankfull depth and width for the pre-disturbance river Cherwell.

VARSLOPE simulations were undertaken at specific sites in the restoration reaches defined previously. Accordingly, it was necessary to define the values of model input parameters at these locations. Most parameter values were defined in relation to direct physical measurements (e.g., bed-material grain size, bank material characteristics), but others require certain assumptions to be made, thereby introducing additional uncertainty into the analysis both via *unreliability uncertainties* based on the values used in the estimations, and structural uncertainties that are inherent (and often unspecified) in the methods adopted. For example the bankfull discharge is commonly assumed to be a formative discharge with a recurrence interval (RI) in the range of 1–2 years (Wolman and Leopold, 1957). In this study, we used a 1.5-year RI to define the formative flow, though we have no means of verifying that this value is appropriate. Furthermore, no data were available to define the Cherwell's sediment load in the study reaches. We estimated this parameter by constructing an empirical relationship between flow discharge and sediment load using data from Hey and Thorne (1986). The Hey and Thorne (1986) data are based on the physical characteristics of 62 British gravel-bed rivers, but we restricted our analysis to a subset of the data for rivers with similar gradients (< 0.0015) to the Cherwell. Certain assumptions are also implicit in our use of well-established empirical–physical relationships (Hey, 1979; Clifford et al., 1992) to calculate bed roughness height based on measured bed-material grain sizes (Table 28.1). Finally, we assumed that the bank roughness height was equal to that calculated for the bed. In addition to uncertainty introduced by these assumptions, one input parameter (the critical shear stress for entrainment of bank material, τ_c) is not readily measurable, nor can it easily be estimated, except via model calibration. This was achieved by adjusting τ_c until simulated equilibrium width values agreed with the 'observed' bankfull width derived from the empirical hydraulic geometry analysis undertaken previously (Fig. 28.3).

Since the calibration process forces agreement with 'observed' data, uncertainty associated with each of the individual input parameters described previously is effectively lumped into a single term (τ_c). This is convenient because, if the gross uncertainty in τ_c can be established, the impact of this on simulated depth can readily be determined. Specifically, multiple simulations can be undertaken using a suitable range of τ_c values obtained from calibrations using width values according to the quantified uncertainty. The results can then be used to define a corresponding range of simulated depth values. In a preliminary set of VARSLOPE simulations we determined the uncertainty in τ_c by calibrating the model using a range of channel width values reflecting the (map-derived) error quantified for each site. Subsequently, a second set of simulations was undertaken to evaluate the impacts of uncertainty in τ_c on simulated pre-disturbance depth values (for complete reporting, see Sear et al., 2001).

Based on the preceding analysis, pre-disturbance channel dimensions of the river Cherwell can be reconstructed as a function of drainage area and visualised in relation to the associated degree of *unreliability uncertainty*. The relative uncertainty is greater for the reconstructed width ($\pm$39– $\pm$24%) than the depth ($\pm$28–$\pm$20%), but for both variables the uncertainty is scale-dependent, declining in the stated ranges as drainage area increases from 150 to 800 km^2 (Sear et al., 2001). This implies that *unreliability uncertainty* (e.g., *inaccuracy*) is not in this case amplified as a result of using uncertain channel width data in the calibration of the VARSLOPE model. The decline in relative uncertainty as a function of increasing width and depth as drainage area increases is explained by recalling that uncertainty in this case study is essentially derived from the fixed ($\pm$3.25 m) error involved in estimating channel width from the map data. These, admittedly, specific types of uncertainties directly influence specification of the designed channel dimensions. One can visualise the uncertainty reported in Fig. 28.3 as the dimensions of a cross-section with a vaguely defined bed and bank boundary. The significance of this vaguely sized channel can be expressed simply as a comparison between floodplain reconnection and downstream flooding at the two extremes of channel capacity resulting from the width and depth uncertainty (e.g., smallest width and depth combination versus largest width and depth combination). Were the aim of the Cherwell restoration project habitat enhancement instead of floodplain reconnection and flood defence, the significance would be expressed in terms of influence on physical habitat. Depending on the metric of significance deemed important, the same uncertainty (in this case channel dimensions) could be essential to project success, or have virtually no significance.

4. Contrasting philosophies towards uncertainty

Having established that uncertainties are prevalent in restoration and that their significance varies depending on the context, we are still left with a philosophical choice as to how to approach uncertainty. There are at least five philosophical approaches towards uncertainty (Wheaton et al., in press):

- *Ignore Uncertainty*: A passive approach to uncertainty that works if uncertainty turns out to be insignificant. Uncertainty might be ignored if it has been proven to be insignificant. More often, uncertainty is ignored because we are ignorant to it or aware of it but do not have the time, resources or know-how to deal with it actively.
- *Eliminate Uncertainty*: Only considers a narrow definition of uncertainty and assumes that an absolute answer exists to the thing(s) we wish to know (e.g., someone's name is an example of a something that uncertainty can be completely eliminated about). Under broader definitions of uncertainty (e.g., Van Asselt and Rotmans, 2002), this approach is typically impossible. The philosophy is based on a positivist view of uncertainty as an absolute lack of knowledge and/or sign of weakness.
- *Reduce Uncertainty*: This is a prevalent approach amongst reductionist scientists, in which uncertainty is viewed as an unfortunate characteristic of the things we wish to know about. Under the Van Asselt and Rotmans (2002) typology, this approach only makes sense for *unreliability uncertainties* (e.g., measurement errors). The approach accepts that some uncertainty will always be present, but strives to reduce it to an absolute minimum.
- *Cope with Uncertainty*: This is a slightly more practical adaptation of the *reduce uncertainty* philosophy, in which there is a fuller admission of the prevalence of uncertainty. The approach, therefore, only seeks to *reduce uncertainty* where it is practical to, and learn to adopt ways of coping with uncertainty otherwise. The approach still fundamentally views uncertainty as a negative thing.
- *Embrace Uncertainty*: This approach only makes sense if a very broad definition of uncertainty is accepted (e.g., Van Asselt and Rotmans, 2002). Uncertainty is viewed without contempt or admiration, and it is recognised that it sometimes results in positive outcomes (e.g., surprise, unforeseen benefits) and sometimes results in negative outcomes. Uncertainty is *embraced* in that it is accepted for what it is – information. In only extreme cases is uncertainty a complete lack of knowledge (e.g., *irreducible ignorance*), and in most cases uncertainty is thought to be transformable into uncertainties of lesser magnitude that provide more information.

The overlap between these five philosophical strategies is illustrated in Fig. 28.4. To 'ignore' uncertainty is the most basic of the philosophical strategies towards uncertainty, and the one most commonly exercised by the restoration community (Wheaton et al., 2006). The appropriate choice of approach depends very much on the context of the restoration approach, the significance of the uncertainty under consideration and the adopted definition of uncertainty. Regardless of the approach, the significance of essential uncertainties should be assessed and more clearly communicated to the restoration community. Efforts based strictly on *reducing uncertainty* demand enhanced levels of monitoring and site investigation to establish causation, but these are likely to be frustrated by *uncertainty due to variability*. In the Cherwell example, the level of uncertainty was quantified and the sources constrained. Approaches based

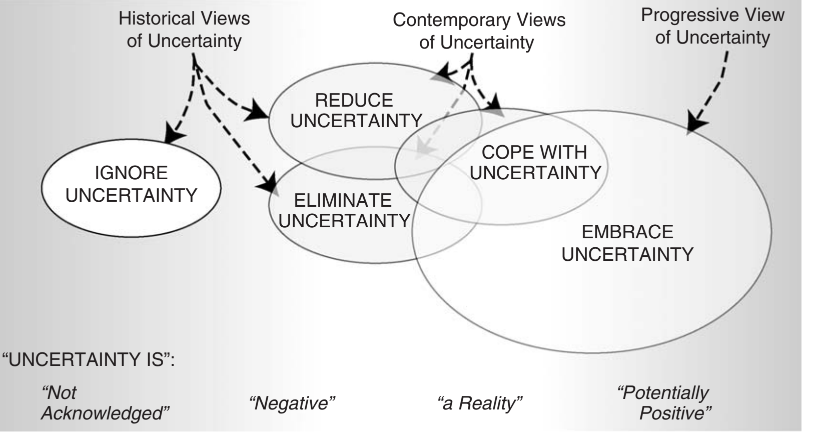

Figure 28.4. Five philosophical attitudes towards uncertainty. The Venn diagram is meant to illustrate the overlap between contemporary attitudes towards uncertainty. Note that ignoring uncertainty, shares no overlap with contemporary attitudes towards uncertainty. (Figure taken from Wheaton, 2004.)

on further constraining uncertainty will again require more investment in pre-project research and investigation. Ignoring the uncertainty may invalidate the model output, which in the Cherwell example could have implications for the understanding and management of downstream flooding. If the flooding is acceptable, a more relaxed approach to dealing with the uncertainty may be warranted. Incorporating uncertainty specifically within a modelling context is an established method for *coping with uncertainty*, and several approaches exist to support this; for example GLUE (Beven, 1996; Brazier et al., 2000) or Monte Carlo simulation (Binley et al., 1991).

In almost any restoration example, the uncertainty in the project outcome (e.g., *uncertainty due to natural variability* arising from subsequent channel adjustment) will be critical in determining whether the project is cast as a failure or a success. As stressed in the introduction to this paper, an unforeseen project outcome may be identified by stakeholders as project failure, rather than success (Kondolf, 1998). Here it is helpful to revisit this concept of plausible outcome space. In Fig. 28.5 all outcomes are defined within the space bounded by the four shaded boxes. The boxes represent the four variants of outcomes based on desired versus undesired outcomes and unforeseen benefits versus unforeseen consequences. Notice that the plausible outcome space only occupies a smaller subset (represented here with a circle) of the larger outcome space. If we do nothing other than estimate the range of plausible outcome space, our uncertainty is contained within this space. If we fail to communicate our uncertainty, we risk encouraging the incorrect presumptions that the target state will certainly be achieved or that our uncertainty occupies the entire output space.

Within the restoration context, perhaps the most promising and practical philosophy towards uncertainty is to embrace it. Wheaton (2004) details the many reasons for this; but one of the most compelling is that under a broad and holistic consideration of uncertainty, it can be quite a positive thing (Clark, 2002; Lempert et al., 2003). From an engineering perspective it may be disappointing that our simulation models cannot make predictions with absolute certainty. However, uncertainty in

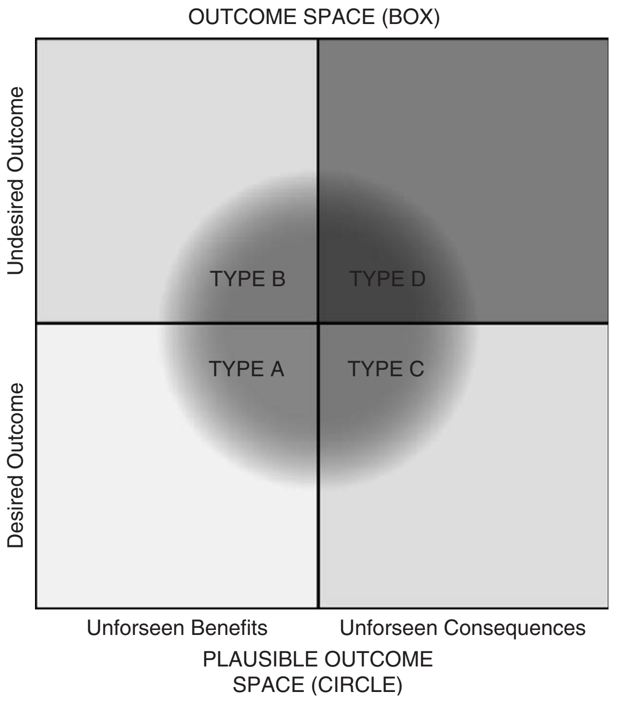

Figure 28.5. A mapping of the plausible outcome space.

defining future scenarios to drive such models can explain why this is the case (Lempert et al., 2003). From a practical stand point, systems that exhibit natural variability in their processes are likely to be more self-sustaining and resilient (Bilbly et al., 2003; Wissmar et al., 2003). Furthermore, an open and transparent communication of uncertainties in restoration is likely to lead to more realistic expectations about the potential outcomes of restoration projects (Wheaton, 2004). The lack of an adaptive management approach to river management is likely to impede such communication (Newson and Clark, in press).

5. Where does embracing uncertainty take us?

An interesting question arises about whether embracing uncertainty necessarily leads to more complicated explanations. The two case studies we have provided seem to suggest to the reader that the answer is yes. However, when the significance of uncertainty is cast in a slightly different context, we can find examples where embracing uncertainty need not necessarily be equated to seeking more complicated models (whether they are conceptual or mathematical).

If we are willing to accept a broader range of plausible outcomes (e.g., a larger diameter circle in Fig. 28.5) we can adopt a simpler approach to embracing uncertainty…we live with it! This is different to ignoring it. The expectation based on ignoring uncertainty is currently that we expect the restoration to attain its prescribed target (e.g., type A or C in Fig. 28.5) – if it does not (e.g., type B or D in Fig. 28.5) then the restoration is a failure and unforeseen additional management will be necessary. The expectation based on embracing uncertainty would be that we would hope that the prescribed target is attained, but we fully accept that it might not be. In this case because we are explicit about the significance of uncertainty, not attaining the initial target is not failure of the project but attainment of one of the other plausible end states. Other plausible end states can include both desirable outcomes and undesirable outcomes. However, it is important to recognise that unforeseen benefits can arise from what initially might have been perceived as undesirable outcomes (e.g., type B in Fig. 28.5). Under a model of restoration wherein uncertainty is ignored, such undesirable outcomes would have been written off as failures. The benefits of such failures are only recognised through monitoring and adaptively managing on this information. Conversely, unforeseen consequences can arise from what initially might have been perceived as a desirable outcome (e.g., type C in Fig. 28.5). Under a model that ignores uncertainty, such consequences would have gone undetected.

To return to the example of spawning habitat; Wheaton et al. (2004) documented an unforeseen consequence in a project that met its restoration objectives and was assumed to be a success. Rehabilitated spawning beds downstream of a reservoir were constructed according to a design specification and shown through monitoring and habitat suitability modelling (considering hydraulics) to be providing suitable spawning habitat. However, a few years after construction an unforeseen consequence of reservoir and hatchery operations was resulting in the smothering of the spawning beds with invasive growth of aquatic vegetation. This growth was not incorporated in the original modelling that supported the design. In an embracing uncertainty framework, this scenario might have been identified specifically as one of the plausible outcomes. Even if it were not foreseen, the adaptive management response focuses attention on the newly acquired understanding of the role of vegetation, rather than locking the managers into a programme of vegetation maintenance in order to achieve a static target. Under an ignoring uncertainty approach, all the valuable information gleaned from the restoration project, the high-quality spawning habitat provided in the early stages of the project's lifetime, and the potential for higher quality habitat to return would be brushed aside and the project labelled as a failure.

The acceptance of a range of plausible outcomes to a restoration project rather than a single target end state is supported by notions of fluvial systems as dynamic in the face of changing external and internal drivers (climate change, land cover change tectonic activity etc.). Indeed, geomorphology is replete with examples and conceptual models that demonstrate more or less sensitivity to changes in driver variables. Furthermore, the historical record of river system response is seldom static. Moreover, ecosystem models recognise the value and importance of disturbance as one of the main drivers of biodiversity and functionality. Thus by embracing uncertainty, restoration projects would incorporate dynamic systems whilst recognising that a range of plausible outcomes exist that may be more or less beneficial to ecosystem functioning, and which may have a range of consequences.

An important element to this discussion is the consideration of spatial scale. The plausible outcome space in a restoration project is strongly related to a location.

Most restoration projects are small-scale, reach-based schemes, nested within a wider, changing catchment. Frameworks that embrace uncertainty need to incorporate the spatial dimension; for example the attainment of a desirable outcome may occur, but not within the physical space that was the initial focus of the restoration (e.g., the local community may benefit rather than the river per se). Similarly, it might be possible to attain the desirable outcome elsewhere in the system rather than at that specific location.

As the requirements of restoration projects become more sophisticated and include, for arguments sake, a definition of habitat disturbance frequency and type; then the mapping of the plausible outcome space may need to be more refined. The demands of the restoration science community then becomes one of providing methods for determining the plausible outcome space for a given set of boundary conditions and restoration design (i.e., fluvial geomorphology). A further caveat is that these need to be spatial and capable of working over the longer timescales associated with ecosystem restoration. One approach being explored is to utilise the growing field of landscape evolution modelling (LEM), and explicitly adapt it for the forecasting of plausible outcomes from single or multiple restoration projects within a catchment. Recent advances in landscape scale modelling are providing opportunities to integrate hydrological, land use history and geomorphology with models of vegetation succession (Richards et al., 2002). Coulthard and Macklin (2001) report such an exercise for a medium-sized basin in the UK using the (cellular automaton evolutionary slope and river (CAESAR) model. Such models offer the opportunity for river managers and stakeholders to engage with concepts of dynamic natural environments such as complex response, and enable informed discussion and communication of the significance of geomorphic adjustment over longer timescales (Sear and Arnell, 2006). Furthermore, spatially distributed models of hydrology and sediment flux share commonalities with landscape ecology models of patch dynamics. These provide a basis for more effective long-term simulation of ecosystem processes.

We are currently refining the CAESAR model to include an ecohydraulic model within it and using the simulations to explore the plausible outcome space of restoration activities across a case study catchment in California's coast range. The process of modelling individual simulations that make predictions of a single plausible outcome is modularly implemented and diagrammed in Fig. 28.6. The modular flexibility of the simulation approach allows process and scenario variants to be used to develop many individual simulations whose outputs can be collectively analysed to explore the plausible outcome space. While such an intensive modelling effort may not be necessary or feasible in all restoration instances, it is being pursued here for two primary reasons. First, in situations where we would like to narrow down the possibility of a type D outcome and better understand the triggers and controls that lead to a type D outcome, scenario modelling shows some promise. Second, even though we might know that the uncertainty in restoration outcome is large (i.e., the plausible outcome space is a bigger circle) we really do not yet understand the significance or ramifications of the uncertainty. Where fish, bank erosion and flooding are the concerns, an LEM coupled to a fish habitat model can allow us to explicitly explore the significance of the uncertainty.

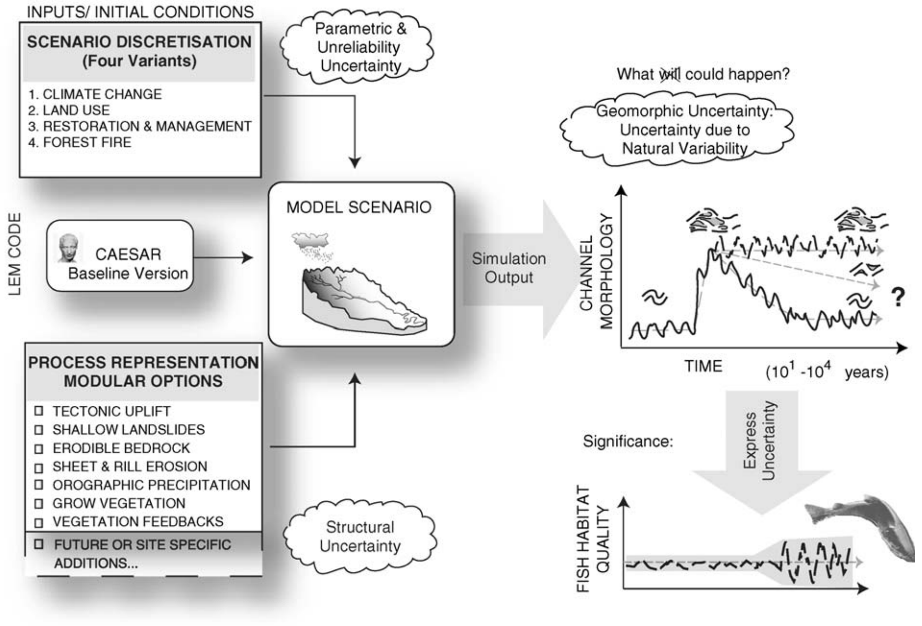

Figure 28.6. An example of a landscape evolution modelling approach being used under the embracing uncertainty philosophy to address geomorphic uncertainties in river restoration.

6. Conclusions

This paper has argued that the increasingly complex goals and targets set by the river restoration community of scientists and practitioners necessitate greater consideration of uncertainty. To date limited recognition of the uncertainties involved and certainly limited uptake in restoration planning (with notable exceptions) has been the practice. However, with the results of science-based monitoring projects beginning to question the validity of the restoration process as currently practiced, the significance of uncertainty is becoming more apparent. Uncertainty is often crudely defined, thus a fundamental step is to structure the sources of uncertainty within river restoration. The typology of Van Asselt (2000) is advanced as a suitable framework for understanding and identifying the sources of uncertainty within river restoration.

The significance of uncertainty is explored for two areas of the restoration process. The uncertainty in the conceptual models used to support the development of restoration strategies are shown to result in the failure to treat the actual source of the biological problem. In another example, the choice of conceptual models available leads to significant design uncertainty. For some sources of uncertainty, it is possible to estimate the magnitude. This is undertaken for a typical restoration design problem using standard geomorphological modelling approaches. The resulting uncertainty around the estimation of pre-disturbance bankfull channel dimensions range from 20 to 39%.

It is demonstrated that uncertainty in restoration exists in a variety of forms that influence the whole of the restoration process; but these can be identified. The question remains how to respond to them. We argue that it is no longer appropriate to ignore uncertainty, and, through recognition of the multiple types and sources uncertainty it is impossible to constrain. We make a case instead for a route that seeks to embrace uncertainty, and we demonstrated both complicated and simplified examples of such. We also briefly highlighted an active area of research that is seeking to embrace uncertainty using coupled LEM and habitat modelling. All embracing uncertainty approaches are implicitly linked to adaptive management approaches. These recognise from the outset that river systems are inherently dynamic and that it is the operation of these dynamics that form the basis for any restoration. Although absolute prediction of restoration outcomes is impossible due to uncertainty, scenario modelling might be used to gain new insight into what restoration may mean. The hope is that ultimately this will help decision-makers, restoration practitioners and stakeholders form more realistic expectations about restoration, as well as furthering the scientific understanding of river response in a changing world.

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Discussion by Gary Williams

I very much agree with the need to consider uncertainty and present information about risks. Life is always uncertain and people do understand this. We should be presenting information in a way that makes the risks known. In many fields, not just in the gravel-bed rivers fraternity, people are grappling with risk, and developing ways of analysing and presenting information on risk. I see this paper as part of that process of risk analysis and the development of formal procedures for dealing with uncertainties.

Discussion by Gordon Grant

With river restoration emerging as an international focus of both river management agencies and scientists, it is interesting to note the different disciplinary ‘‘cultures’’ driving restoration in different parts of the world. One can view this perhaps as a ternary diagram with the three major disciplines involved – engineering, ecology and