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Title Risks and Uncertainty in River Rehabilitation

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Summary

Analyses of costs and benefits require the prediction of the effects of restoration measures and the quantification of societal values. Both of these estimates are uncertain. In this report, some of the key issues related to the assessment, description and quantification of uncertainty are discussed and guidelines are provided for considering uncertainty. This report provides a brief overview on the representation and quantification of uncertainty in scientific prediction followed by examples of typical risks associated with river restoration that could lead to unintended, adverse effects and in more detail, how uncertainty can be considered in CEA/CBA and in MCDA.

There are two important **sources of uncertainty** to consider in environmental management in general, and in particular for river restoration:

- **Uncertainty about scientific predictions of outcomes.**
Depending on alternatives, this requires prediction and uncertainty estimation of the behavior of a natural system, natural-technical system, or even of a combined natural-technical-socio-economic system (e.g. in case of measures that include incentives to some of the affected stakeholders). In particular, one has to consider the potential for adverse outcomes as discussed in chapter 3.
- **Uncertainty about the preferences of the society elicited from inquiries or stakeholders.**
In addition to the difficulties of the stakeholders to be aware of their own preferences and to be able to quantify them, this also includes their risk attitude (how uncertainty about the outcomes affects their preferences).

Policy recommendations:

- **Communication of uncertainty is a key element of any communication of scientific predictions.** Visualization of uncertainty ranges can support this task. Lack of communication of scientific uncertainty in the past led to a reduction of trust of stakeholders to scientists.
- **Clearly separating scientific predictions and societal valuations is an essential element of any decision support procedure.** Uncertainties in both elements should be clearly communicated separately. In particular if there are disagreements among experts about scientific predictions and of stakeholder groups about preferences.
- **Uncertainty about scientific predictions can be addressed by probability distributions and scenarios; uncertainty about societal preferences are often better addressed by sensitivity analyses** of the ranking of the alternatives resulting from combining predictions of the outcomes of decision alternatives with preferences.

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1. Introduction

The EU Framework Program 7 funded project Restoring Rivers for Effective Catchment Management (REFORM) aims to develop guidance and tools to ensure river restoration measures are cost-effective and support future River Basin Management Plans (RMBPs) for the European Water Framework Directive (WFD). Formally, this can be supported by cost-effectiveness analysis (CEA), cost-benefit analysis (CBA), or multi-criteria decision analysis (MCDA), as described in the deliverable 5.2 (Brouwer et al. 2015). These analyses require the prediction of the effects of restoration measures and the quantification of societal values. Both of these estimates are (very) uncertain.

In this report, some of the key issues related to the assessment, description and quantification of uncertainty are discussed and guidelines are provided for considering uncertainty when using these decision aids. Often, there is uncertainty in the precise quantification of required costs and achieved benefits of rehabilitation measures. However, there may even be adverse effects. To account for these problems, we address both quantification and consideration of uncertainty in general and specific circumstances that may even lead to adverse effects.

The remainder of this report is structured as follows. In chapter 2 a brief overview is provided about the representation and quantification of uncertainty in scientific prediction. In chapter 3 typical risks that could lead to unintended, adverse effects, are discussed. In the chapters 4 and 5, it is described in more detail, how uncertainty can be considered in CEA/CBA and in MCDA, respectively. Finally, conclusions and policy recommendations are provided in chapter 6.

2. Uncertainty in Scientific Prediction

By Peter Reichert, Eawag, CH

Environmental management must be based on the best available scientific knowledge. This is mainly needed to predict the outcomes of decision alternatives which are required to evaluate the alternatives with the societal preferences. As scientific knowledge is always uncertain, a necessary element of scientific prediction is the assessment of uncertainty.

In this chapter we will discuss the mathematical representation of scientific knowledge, its acquisition, and how to use it for scientific prediction. This section is primarily based on section 2 of the review written by Reichert et al. (2015).

2.1. Representation of Scientific Knowledge

The philosophical basis of scientific reasoning is linked to the discussion of the “correct” interpretation of probability (see Hájek 2012 and references therein or Chalmers 1999 for a broader coverage of the philosophy of science).

In the context of using probabilities to describe scientific knowledge, we emphasize the intersubjective interpretation (Gillies 1991, Gillies 2000, Reichert et al. 2015). In this interpretation, probability distributions are used to characterize the knowledge of the scientific community about attributes characterizing the system under investigation. With intersubjective probabilities we mean that the probability distributions should reflect the joint knowledge of a representative group of scientists. This can either be achieved by the agreement of a group of scientists about their joint belief or by aggregating individual beliefs. The concept of intersubjective probabilities is in concordance with scientific quality control in science, in particular, with peer review, in which multiple experts have to agree about the techniques and conclusions of a scientific study (Bornmann and Daniel 2010). Intersubjective probabilities are contrasted with objective probabilities that characterize the material world independently of humans (the most important one being frequentist probabilities that characterize the limit of frequency distributions achieved by repeating an experiment that has some random components) and by subjective probabilities that describe individual beliefs. All of these interpretations are needed in some contexts, but intersubjective probabilities seem the most adequate description of scientific knowledge (see Reichert et al. 2015 for a more extensive discussion).

In practice, we may have to account for disagreements between scientists about a probability distribution representing the current state of scientific knowledge. An option of considering such ambiguity is to describe the knowledge by a set of probability distributions rather than a single distribution. Such sets are also known as imprecise probabilities (Walley 1991; Rinderknecht et al. 2012b; <http://www.sipta.org>). When used as prior distributions in Bayesian inference, this leads to so-called robust Bayesian analysis (Berger 1984; Berger 1994; Pericchi and Walley 1991; Rinderknecht et al. 2014).

In cases in which imprecise probabilities indicate a too high degree of ambiguity (Rinderknecht et al. 2012b), probabilistic descriptions can be combined with alternative future scenarios (Schoemaker 1995; Ringland 2006), e.g. for the description of the

development of external influence factors or driving factors. Probability distributions are then formulated as conditional probabilities given these scenarios.

Alternative mathematical frameworks have been suggested to describe uncertain (scientific) knowledge (Dempster 1967; Shafer 1976; Zadeh 1978; Moore 1979; Dubois and Prade 1988; Helton and Oberkampf 2004; Dubois 2006; Colyvan 2008). We suggest to consider the concerns with precise probabilities that led to the development of these theories by using precise probabilities in cases with few ambiguity, imprecise probabilities in cases with significant ambiguity, and combine these approaches with future scenarios where appropriate. For characterizing the involved distributions and scenarios in the context of describing scientific knowledge, the intersubjective approach is essential (Reichert et al. 2015).

2.2. Acquisition of Scientific Knowledge

Scientific knowledge for environmental decision support is usually elicited from experts (Morgan and Henrion 1990; Meyer and Booker 2001; O'Hagan et al. 2006). Intersubjective probabilities can either be obtained by eliciting from a group of experts or by aggregating probability distributions of individual experts (Winkler, 1968; French, 1985; Genest and Zidek, 1986; Clemen, 1989; Clemen and Winkler, 1999). The linear opinion pool (Stone 1961) aggregates distributions by taking the weighted average of individual distributions. This technique has reasonable properties (in particular, it considers that the uncertainty does not decrease by asking more experts who agree on the current state of knowledge), it is easy to understand, and there is empirical evidence that it is successful (Clemen 1989).

2.3. Getting Scientific Predictions

There are essentially two options to get predictions of the outcomes of decision alternatives:

1. We can get conditional predictions of the outcomes of all decision alternatives given their definition and scenarios of driving forces.
2. We can acquire knowledge in the form of a mechanistic model that makes it possible to predict the outcomes by propagating the inputs defined by decision alternatives and scenarios to the output.

Option 1 can be implemented more quickly, but option 2 is potentially more universal. Depending on the available knowledge and resources one or the other approach, or a combination of both, may be appropriate for a given decision problem.

3. Risks of Restoration Projects

In this chapter, we describe a set of risk of restoration projects that could even lead to unintended, adverse effects. It is important to keep such adverse effects in mind, as uncertainty analyses based on a too narrow scope may fail to identify these potential problems.

We focus in this section on potential adverse effects occurring in established rehabilitated reaches of rivers. In addition, adverse effects of the construction process, such as resuspension of sediments, mobilisation of deposited pollutants or eradication of rare species, should also be kept in mind.

The following table provides an overview of which potential measure groups (according to the REFORM WIKI, see <http://www.reformrivers.eu>) may be particularly susceptible to which risks:

	3.1. Increased Flooding Frequency	3.2. Gravel Retention	3.3. Excessive Bank Erosion	3.4. Problems of Navigation	3.5. Pollution of Groundwater used for Drinking Water	3.6. Water Temperature Increase	3.7. Increased Spreading Rates of Invasive Species	3.8. Riparian Vegetation Encroachment	3.9. Increasing Habitats for Mosquitos	3.10. Increasing Accident Rates of Visitors
Water flow quantity improvement	x		x		x					x
Sediment flow quantity improvement	x			x					x	
Flow dynamics (both water and sediment) improvement	x		x	x	x					x
Longitudinal connectivity/continuity improvement	x						x			
River bed depth and width variation improvement	x	x	x	x	x	x		x	x	x
In-channel structure and substrate improvement	x	x	x	x	x			x	x	x
Riparian zones improvement			x					x	x	x
Floodplains/off-channel/lateral connectivity habitats improvement	x		x	x	x	x		x	x	x

3.1. Increased Flooding Frequency

By Michelle Smith and Natalie Angelopoulos, UHULL, UK

Problem description

In freshwater ecosystems, climate change will influence changes in precipitation and runoff further affecting the average discharge, timing, duration and inter-annual variability of peak and low flows. This intensification in variability of extreme flood events are now widely recognized as a major challenge for flood risk management and as a consequence, pressures from flood protection activities are predicted to intensify in the future. In addition to climate change, river rehabilitation measures can increase flood frequency. Instream rehabilitation measures will alter the hydraulics (roughness, capacity, turbulence, flow pattern) of the channel and impact on the sediment dynamics and frequency of out of bank flows (Janes et al., 2005). Examples include cobble riffles, rock weirs and cascades having a medium to high risk of increased flood levels with greater potential to affect conveyance and roughness (Janes et al., 2005). Placement of large woody debris has the potential to affect flood levels but risk is related to the extent of the works (Janes et al., 2005). Unless woody debris is blocking more than 10% of the cross-sectional area of a river it is unlikely to impact on water levels and therefore should not be removed (Water and Rivers Commission, 2000 in <http://evidence.environment-agency.gov.uk/FCERM/en/SC060065/MeasuresList/M5/M5T3.aspx?pagenum=2>). Narrowing of the channel has the potential to affect conveyance and flood levels but if the narrowing is local then the rise may be minimal, a few millimetres in a flood situation.

Counter-measures

Nature based restoration is a counter measure and means working with natural processes, including managing flood risk by restoring and emulating the natural regulating function of catchments, rivers and floodplains. The application of natural flood defence strategies is a nature based solution that integrates the objectives of the FD with the objectives of the WFD. Strategies to combine flood protection and ecological restoration like ECO-Flood have been developed and promote the use of floodplains as natural flood defence measures, while at the same time optimising other compatible functions and values through conservation and restoration (Blackwell & Maltby 2006), creating multiple benefits across social, environmental and economic dimensions. In general, natural flood risk reduction measures aim to increase the retention capacity of upland area storage capacity of floodplains or discharge capacity of river channels. A variety of natural flood defence measures have been applied in projects across Europe, for example, natural measures involve using land to temporarily store flood water away from high risk areas, reconnecting rivers to their floodplains, restoring degraded peat bogs or blocking artificial drainage channels, lengthening watercourses to a more natural alignment, reforesting floodplains will also help to slow run-off and increase infiltration (Environment Agency 2010).

Unfortunately nature based solution are not always suitable, especially to reduce flood risk in areas where the surrounding land use cannot be flooded, for example urban areas and some agricultural land. Here, careful consideration is needed when placing instream

structures, for example large woody debris used in complex placements that mimic natural conditions tend to be more stable because they have greater flexibility to adapt to changing channel and flow conditions (Abbe and Montgomery, 1996 in <http://www.Robin-wood.it>) and placement should avoid sites where the washout of debris could block downstream bridges and culverts (www.Robin-wood.it). Regular maintenance of willow spiling can keep it under control (Janes et al., 2005). If narrowing of the channel is not supported by flood risk management, other measures used in combination could be used to offset the narrowing for example, re-profiling or provision of alternative flood storage elsewhere (Janes et al., 2005). Hydraulic modelling is recommended to determine the effect of single, or a suite of, techniques/structures/measures on the river (Janes et al., 2005).

Case studies

Several case studies demonstrate the multiple benefits that can be gained when incorporating river restoration into flood risk management. Grote Noordwaard, a good example where water retention capacity of the river Boven-Merwede was improved to reduce flooding by creating a large polder. This serves as a flood retention area during river peak flow and also as a nature reserve areas. In this project, agricultural land is given back to the river to improve the discharge during flooding and the nature is allowed to develop. The measures of this project are still in progress, and will be finished in 2016 but it is expected that the measures will have a positive effect on the ecological value of the Grote Noordwaard and the Biesbosch as well as improvement of the flood protection (Angelopoulos et al. 2015). Floodplain excavation at Grensmaas (Border Maas) is another good example where large scale excavation of floodplains by means of gravel extraction, widening of main channel and construction of secondary side channels reduce the risk of flooding whilst enhancing the ecological aspects of the surrounding environment.

<http://www.ecrr.org/RiverRestoration/Floodriskmanagement/tabid/2615/Default.aspx>

<http://evidence.environment-agency.gov.uk/FCERM/en/SC060065/MeasuresList/M5/M5T3.aspx?pagenum=2>

<http://www.Robin-wood.it> The Robinwood Robinflood report: Evaluation of Large Woody Debris in Watercourses

3.2. Gravel Retention

By Diego García de Jalón, UPM, ES

Problem description

Several conditions causes gravel retention in river reaches due to deposition in river widenings, or below engaged logs and woody debris, or weirs can reduce the gravel load downstream. However, retention capacity is small and soon the river recovers its previous sediment yield. This is not the case of large dams.

Vörösmarty et al. (1997) have estimated that more than 40% of global river sediment discharge is intercepted by 633 of the world's largest reservoirs. More recently, Vörösmarty (2003) precise that more than 50% of the basin-scale sediment flux in regulated basins is trapped in artificial impoundments, with discharge-weighted sediment trapping due to large reservoirs of 30%, and an additional 23% trapped by smaller reservoirs.

The presence of major dams and the vast reservoirs of water that they impound alters downstream water flow regime due to dam operations, while the retention of sediments is a largely unavoidable consequence (Braatne et al., 2008).

Retained sediments by dams include the further downstream adjustments of the biota in response to the morphological and bed sediment adjustments that lead to a completely different assemblage of hydraulic and morphological habitats. Petts (1984) adopted a geomorphological approach to build a conceptual model of channel response to large dam construction, which incorporated potential adjustments at different sites along the river below the dam and through time in response to relative (before-after) changes in discharge and sediment delivery, bed material composition, bank erodibility, floodplain connectivity, the influence of unregulated tributaries, and the riparian vegetation.

Predicting the geomorphic response of rivers to the building of large dams is complex but such predictions are needed to mitigate and restore regulated rivers. Grant (2012) provides a recent review of knowledge, which has largely been generated over the last 30 years. Early empirical analysis of observations drawn from multiple rivers by Williams and Wolman (1984) revealed general trends in bed incision depth and its downstream progression below dams. These ideas have been recently updated (Petts and Gurnell, 2005) and also predictive quantitative models have been proposed. For example, Schmidt and Wilcock (2008) determined that river bed incision occurred below dams when the ratio of the pre-dam to post-dam slope needed to transport the quantity and caliber of supplied sediment at the imposed discharge was greater than one, but also, the post-dam flows needed to be competent to transport the post-dam bed sediments (which may be armored) in order to actually incise the bed.

Choi et al. (2005) illustrated the second-order geomorphological impacts of dam construction through a comparison of pre- and post-construction surveys of the Hwang River, Korea. A common morphological response to large dam construction is channel bed incision over a stream length of many km immediately downstream of the dam, as the river erodes its bed to replace sediment trapped by the dam (Ward & Stanford, 2006).

Counter-measures

Maintaining reservoirs in present use and prevent sediment retention is a difficult task. The accumulated sediments must be relocated down into the dam's tail water. This process is called sediment replenishment, and has been implemented in Japan, USA and Switzerland (Cajot et al. 2012). Other effective measures to stop sediment trapping by reservoirs and to decrease the reservoir sedimentation process comprise the construction of sediment bypass tunnels. These tunnels route the sediments (of both bed load and suspended load) around the reservoir into the tailwater during flood events; consequently, sediment accumulation is reduced significantly. Nevertheless the number of realized sediment bypass tunnels in the world is limited (six in Switzerland and five in Japan) due to high investment and maintenance costs. The design of a bypass tunnel (Auel & Boes, 2011) consists of a guiding structure in the reservoir, an intake structure with a gate, mostly a short and steeply sloped acceleration section, a long and smoothly sloped bypass tunnel section, and an outlet structure.

The only efficient measure is dam removal. More than 1,000 dams have been removed across the United States because of safety concerns, sediment buildup, inefficiency or having otherwise outlived usefulness. A recent review of more than 1,000 dam removals in the last 40 years (O'Connor et al. 2015) concludes that rivers are resilient and respond relatively quickly after a dam is removed. Rivers quickly erode sediment accumulated in former reservoirs and redistribute it downstream, commonly returning the river to conditions similar to those prior to impoundment. Studies show that most river channels stabilize within months or years, not decades, particularly when dams are removed rapidly. In many cases, fish and other biological aspects of river ecosystems also respond quickly to dam removal.

Case studies

The Ebro River provides such an illustration at a basin scale. Guillen & Palanques (1992) estimated that the sediment load entering the delta of the Ebro River represented only 1% of the sediment discharge prior to dam construction in the Ebro basin. Before the construction of large reservoirs in the lower Ebro at the end of the 1960s, the sediment transport was estimated to be around $1.0 \times 10^7 \text{ Mt yr}^{-1}$. This amount was reduced to around $0.3 \times 10^6 \text{ Mt yr}^{-1}$ after construction of the dams. Currently, this amount ranges from 0.1 to $0.2 \times 10^6 \text{ Mt yr}^{-1}$, which represents a reduction of more than 99% in sediment transport from natural conditions. On a seasonal scale, the effects of the dams have been the standardization of the river flow and the virtual suppression of peaks in sediment transport.

Another important example of the first-order impacts can be found on the Yangtze River since the closure of the Three Gorges dam. In the period 2003-2006, ~60% of sediment entering the reservoir has been retained (Xu & Milliman, 2008). Although periodic sediment deposition continues downstream of the dam, substantial erosion has also occurred, supplying ~70 million tons per year (Mt.y^{-1}) of channel-derived sediment to the lower reaches of the river. During the extreme drought year 2006, sediment discharge drastically decreased to 9 Mt (only 2% of the 1950-1960s level) because of decreased water discharge and trapping by the dam. Severe channel erosion and a drastic decline in sediment transport have resulted in major pressures on the coastal areas at the mouth of the river and on the East China Sea (Xu and Milliman 2008).

3.3. Excessive Bank Erosion

By Mario Schirmer, Eawag, CH

Problem description

River restoration is intended to enhance river dynamics, environmental heterogeneity and biodiversity, but the underlying processes governing the dynamic changes need to be understood to ensure that restoration projects meet their goals, and adverse effects are prevented. In particular, the danger of excessive bank erosion has to be considered.

Counter-measures

We need to comprehend how hydromorphological variability quantitatively relates to ecosystem functioning and services, biodiversity as well as ground- and surface water quality in restored river corridors (Schneider et al., 2011). This involves (i) physical processes and structural properties, determining erosion and sedimentation, as well as solute and heat transport behavior in surface water and within the subsurface, (ii) biogeochemical processes and characteristics, including the turnover of nutrients and natural water constituents, and (iii) ecological processes and indicators related to biodiversity and ecological functioning. All these aspects are interlinked, requiring an interdisciplinary investigation approach. In some cases, physical measures of bank protection may have to be taken to prevent excessive erosion.

Case study

We investigated and still address these questions in the recently completed RECORD (REstored CORridor Dynamics) and the ongoing RECORD Catchment projects (Schirmer et al., 2014). Our research catchment is the Thur River in Switzerland. The results presented here are from a restored site in Niederneunforn / Altikon where undesired and unexpected erosion took place.

Since 2002, the restored Thur site has experienced large morphological changes triggered by either moderate or extreme flooding events (Fig. 3.1) (Schneider et al., 2011). Moderate erosion at certain river reaches was expected and the development of new gravel bars was desired. The active geomorphodynamics created by the restoration action improved the ecosystem (Schirmer et al., 2014). The larger diversity of habitats provided a broader range of ecological niches thus allowing a higher overall diversity of organisms to colonize the area. However, more active geomorphodynamics may become problematic when excessive erosion takes place (Pasquale et al., 2011). For example, in five years, the gradual formation of a (metastable) point bar on the left river bank has caused the removal of a large fraction of the riparian forest on the opposite bank (Fig. 3.2). The river is now within 20 m of an agricultural field. Hence, strategic balancing between protection and rehabilitation is needed. For this site, now a relatively stable situation has developed. The potential conflicts with land-owners and agricultural use were prevented.

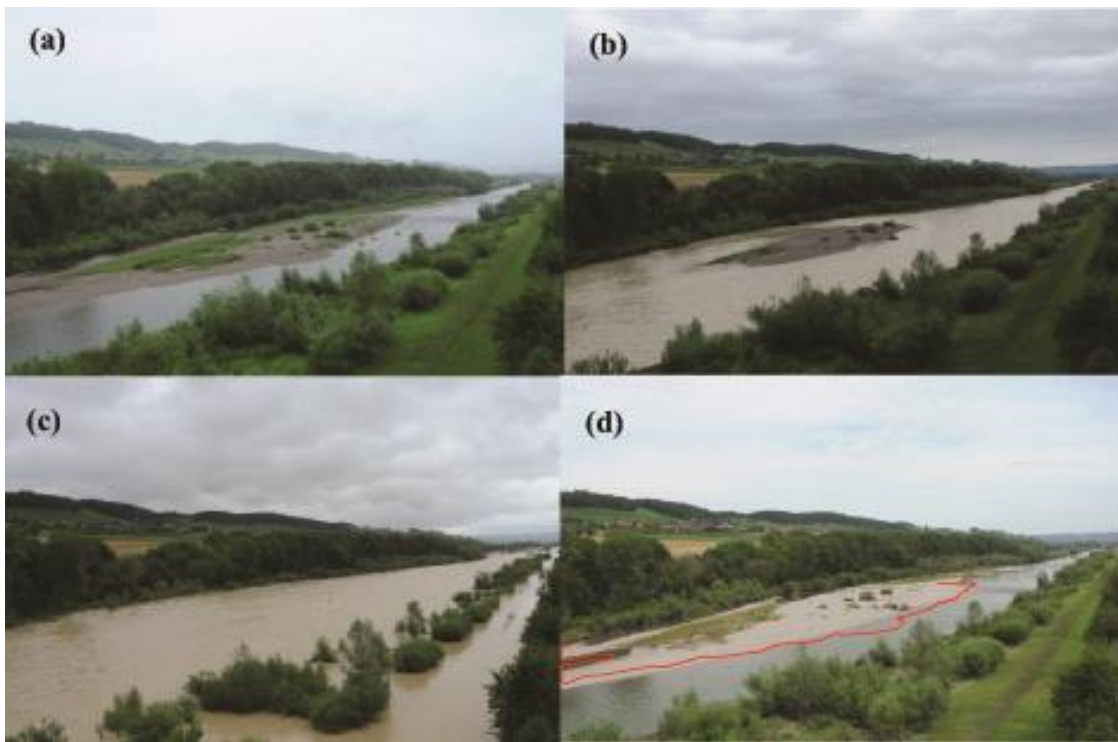


Figure 3.1: Upstream view of the main island of the Swiss River Thur monitored with high-resolution remotely controllable digital cameras (see Pasquale et al., 2011 for details). The sequence (a-d) shows a compilation of the inundation dynamics during the flood in July 2009 (peak flow of 748 m³/s), which resulted in a complete flooding of the restored corridor (c), causing substantial morphologic changes and removal of young vegetation (d). The red contour line in panel (d) shows the comparison with the shoreline of the sediment bar before the flood (panel a) for the same flow rate.

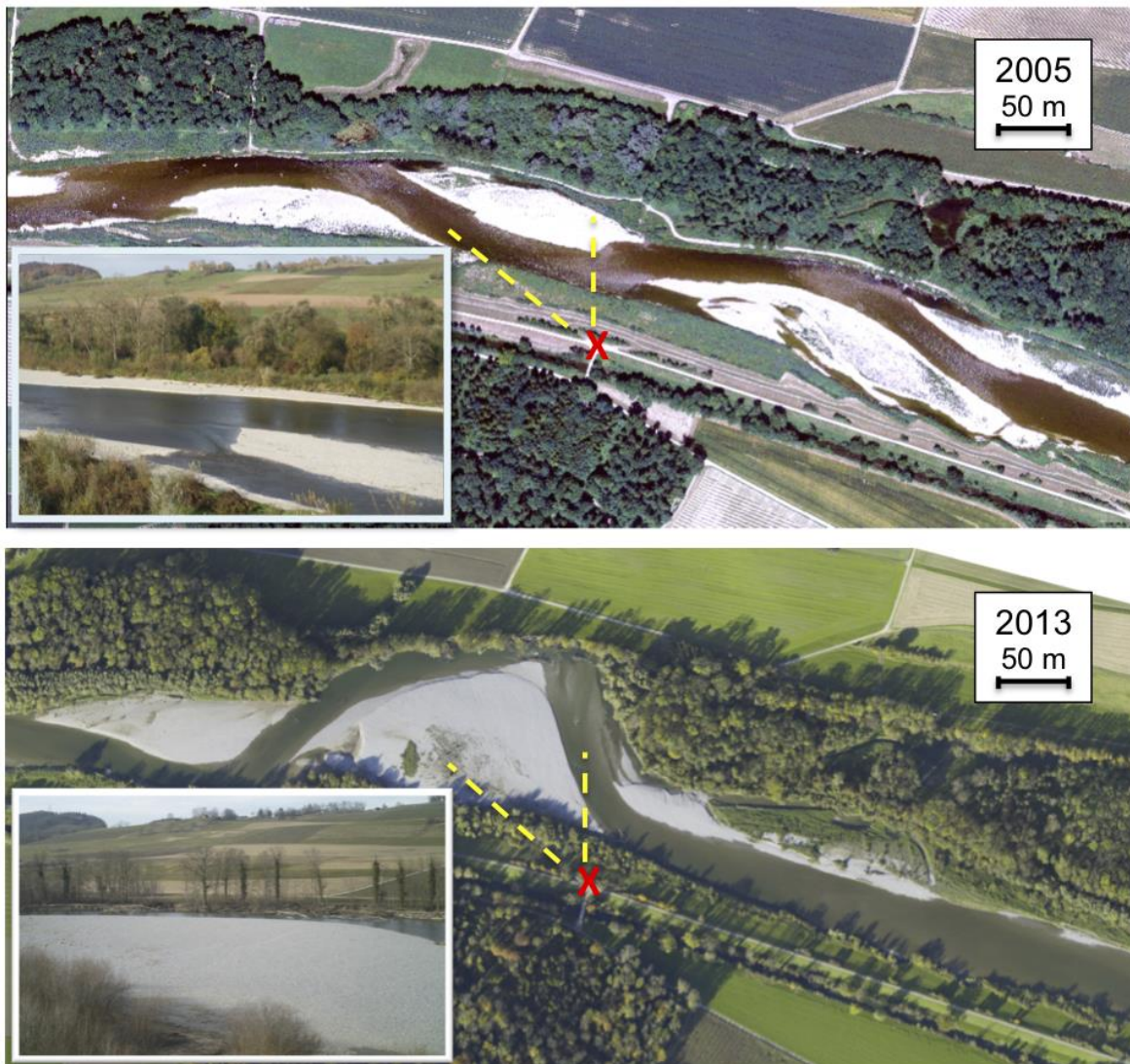


Figure 3.2: Recent erosion: trouble brewing? Starting with the floods of 2010/2011 excessive erosion began in the area pointed out by the yellow dashed lines (river flow is from right to left). Large portions of the riparian forest were removed. The inserted pictures are taken at the locations of the red cross where an observation tower exists (see also Pasquale et al., 2011).

3.4. Problems for Navigation

By Christian Wolter, IGB, DE

Problem description

About 150 years ago, with the beginning of the industrialization period and the rapid development of a railway system as competing mode of transport, a paradigm shift has occurred in inland navigation, from the historical adaptation of numerous vessel types to the specific river conditions to an adaptation of the rivers to continuously increasing vessels by river engineering works. To remain competitive to the railway traffic, vessels had to increase their loads and lower their resting times. This led to the so-called low water regulation of all major European waterways in the second half of the 19th century (Natzschka 1971, Uhlemann 1994, Eckoldt 1998, Rohde 1998). Regulation by dams and wing dikes were the most applied methods to increase fairway depth (Eckoldt 1998, Brolsma 2011, Alexander et al. 2012). Both regulation works either drastically lower the flow velocities or narrow the channel or even both. With increasing river regulation and fairway maintenance, both, the ratio of fairway cross section to channel cross section and the steepness and embankment of the river banks increased. Within the last 150 years the global commercial inland navigation network has been more than 57times enlarged from 11,875 km altered for navigation before 1900 to 671,868 km of inland waterways in 2012 (CIA 2013).

In todays heavily modified, monotonous and homogenised waterways inland navigation and the related fairway maintenance set significant limitations for river rehabilitation (Wolter et al. 2009, Kail & Wolter 2011). If navigation forms a designated use, the existing fairway dimensions have to remain untouched, including the operational water level. This automatically excludes all measures that change depth or width variability (Kail & Wolter 2011). Therefore, the case that rehabilitated river sections can lead to problems for navigation is rather theoretical. In contrast, there is an urgent need to take all the minor and negligible trafficked waterways out of the standard fairway maintenance to open the way for extensive rehabilitation.

Counter-measures

Potential counter measures include the reclassification of waterways to a lower international waterway class. For example, a waterway of the European class VIa has to allow for 95-110 m long and 22.8 m wide vessels with 2.5-4.5 m draught, while the dimensioning vessel for class III waterways is only 67-80 m long and 8.2 m wide with 2.5 m draught and the fairway to maintain correspondingly smaller. Another option would be to designate more one vessel sections with alternating directional traffic. This would reduce the necessary fairway width to the half, because there would be no encounter traffic. Finally, inland navigation could be redesignated as significant use. This would allow for extensive river rehabilitation beyond the limitations of the former fairway dimensions. Such redesignation would not completely abandon commercial inland navigation, but reduce it to vessels adapted to the river. Throughout Europe there are numerous minor waterways with a negligible transport volume compared for example to the River Rhine system that could be redesignated as waterways and ecologically improved to fulfil the requirements of the WFD and the biodiversity strategies.

One measure that can and should be applied in nearly all waterways is fairways adaptation. Today, commercial vessels require much lower fairway width for unhindered navigation due to improved manoeuvrability and navigation assistance systems. Thus, modern ships are able to navigate safely and easily in very much more restricted fairways than traditional ships (Söhngen et al. 2008). That means that the fairways could be signed by buoys substantially narrower and further away from the banks, which would significantly lower the physical forces induced by moving vessels, especially the drawdown, return currents and wake wash at the banks (Söhngen et al. 2008).

Another option would be speed limitation. A moving vessel induces the highest physical forces at the critical speed, i.e. at a vessel speed at that the stern wave start to break. This speed causes the highest waves of more than 1 m height. A reduction of the vessel speed by just 10% will lower the resulting wave height by 50% (Söhngen et al. 2008).

Case studies

So far there are no case studies available, where navigation would have been even slightly limited in favour of river rehabilitation. Instead, rather expensive bank protection has been installed to provide flow and wave protected shallow littoral areas for fish nurseries, aquatic plants and invertebrates in waterways (Wolter 2010, Weber et al. 2012, Rauch 2015).

In Germany, a reclassification of waterways has started based on the water-bound transport volume and frequency in major and minor waterways. In the latter, future maintenance and river engineering work is limited to flood protection only.

In the lower River Havel waterway, Germany, a section of nearly 100 km length has been reclassified from class IV to class I with much lower fairway requirements that opens the room for extensive river rehabilitation (see <https://www.nabu.de/natur-und-landschaft/fluesse/untere-havel/>).

Unfortunately, the highly efficient and easiest to apply measures addressing vessel operation and fairway maintenance are not or only insufficiently (speed control) implemented so far. There is still no example for a fairway adaptation in practise, although this would be a win-win situation for navigation and waterway maintenance. Fig. 3.3 presents an example for the difference between the existing and maintained fairway and the maximum required one for easy navigation. Over many kilometres there are 50 and more metres of fairway dredged than required summing up to tremendous amounts of unnecessary dredged material and costs (Fig. 3.3).

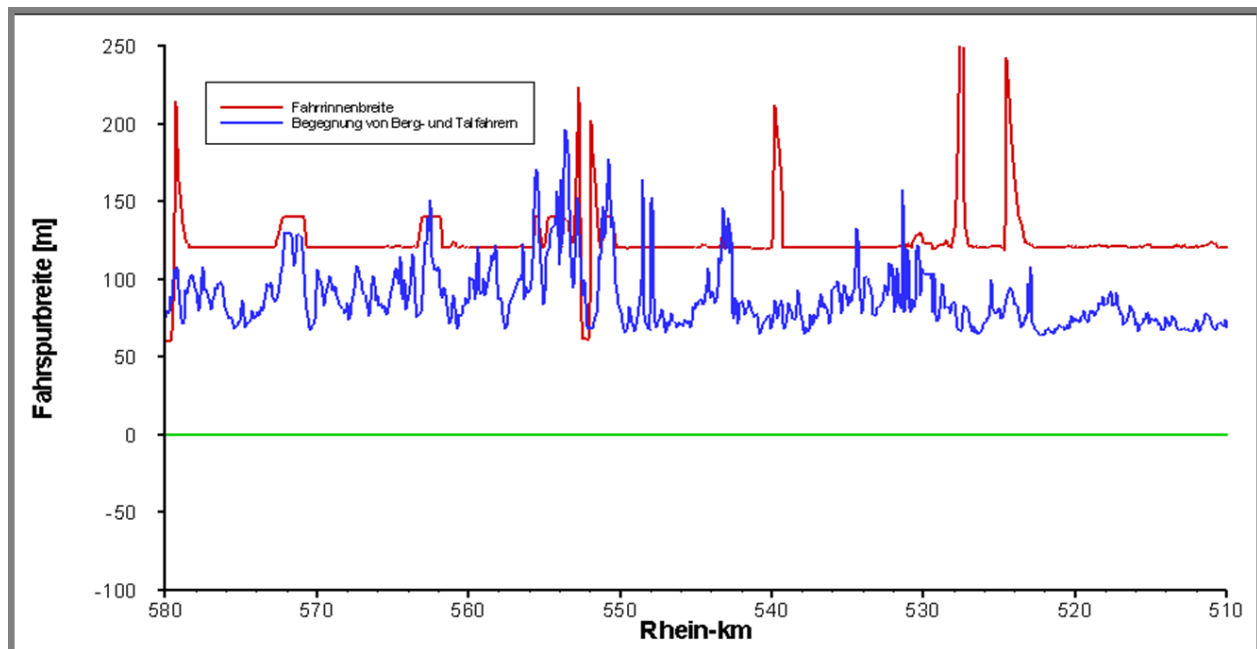


Figure 3.3 Existing (red) and maximum required (blue) fairway width for inland navigation in the River Rhine (from Söhngen et al. 2008).

3.5. Pollution of Groundwater used for Drinking Water Production

By Peter Reichert, Eawag, CH

Problem description

It should be one of the goals of river rehabilitation projects to increase vertical connectivity, as the hyporheic zone is an essential part of the river ecosystem and is required to support its function (Hancock, 2002; Boulton, 2007; Hester and Gooseff, 2010). Restoration projects at least partly achieve this objective (Daniluk et al. 2013). River water infiltrating into groundwater underlies a significant purification process through degradation of organic matter, filtration, and adsorption (Balke and Zhou, 2008; Diem et al. 2013). The decrease in travel time of infiltrated ground water associated with river rehabilitation can decrease this natural purification process and can lead to higher pollution levels of water pumped from the aquifer to be used as drinking water. Thus, there is a conflict between river restoration and drinking water production at wells close to the river.

Counter-measures

Before designing counter measures, it is important to estimate the effect of rehabilitation on stream-groundwater exchange under the actual conditions at the rehabilitation site (Kasahara and Hill, 2008). If the effect is estimated to be unacceptably high, options are not to rehabilitate within the influence zone of groundwater wells used for drinking water

production, moving wells further from the river, or closing wells. This could lead to a considerable increase in rehabilitation costs.

Case studies

Sanon et al. (2012) study the quantification of ecosystem service trade-offs including this conflict between river floodplain restoration and drinking water production in a floodplain of the Danube river near Vienna.

3.6. Water Temperature Increase

By Amael Paillex, Eawag, CH

Problem description

River water is controlled by dynamic energy and hydrological fluxes at the air-water and water-riverbed interfaces (Caissie 2006). Within a given river-floodplain sector, the groundwater input (water-riverbed interface), the topology of the river corridor (air-water interface) can differ in quantity and quality, which is accompanied by differences in water temperature and bio-geochemical parameters (Poole & Berman 2001; Cabezas *et al.* 2011). These exchanges between the interfaces are key parameters that will modify the thermal regime within a river or a floodplain channel (Arscott, Tockner & Ward 2001). A channel or a river with a high water-riverbed connection will experience a stable water temperature among the seasons of the year (Caissie 2006). Oppositely, a river or a channel with a low water-riverbed connection will experience a high water temperature during the summer and a lower temperature during winter in temperate regions (Ward *et al.* 2002). During high water level such as floods, the temperature can temporarily be homogenised among the channels and depends then on the main river channel temperature (Tockner, Malard & Ward 2000). During low water level, the range of thermal differences between waterbodies is expected to be high and constrains the biota by imposing temperature as an abiotic filter and limiting the activity of the organisms (Tockner *et al.* 2000; Poff *et al.* 2006; Bonada, Dolédec & Statzner 2007). Restoration works often change habitat diversity, reduce shadow of trees, increase the lateral connectivity of secondary channels with the main river channel and increase the vertical connectivity of water bodies with the groundwater, for example by dredging the fine sediment of the channels (Stanford *et al.* 1996; Boulton 2007; Jansson, Nilsson & Malmqvist 2007). Restoration efforts like widening the river bed, or reconfiguring the bank shape, or cutting vegetation on the banks are likely to modify the thermal regime of river by changing the water-air exchanges and the water-riverbed interface. Widening a river implies to shallow the water in the river bed, to increase the solar irradiance on water, to reduce water velocity and therefore to potentially increase the water temperature. However, only few studies have been carried out to assess the impact of such operations on the water temperature and its associated effect on the biota (Bernhardt *et al.* 2005; Boulton 2007).

Counter-measures

Replanting trees is a solution to increase shadow on rivers and decrease direct solar irradiance, increasing vertical connectivity with the groundwater is a second solution to reduce the mean annual temperature of water, change channel configuration can also have positive influence on water temperature (Pierce *et al.* 2014). However, nowadays the morphological structure of degraded rivers is in the focus of river restoration programmes with the aim to increase the naturalness of rivers, while restoring cold water fluxes in rivers is less often applied.

Case studies

Studies about the effect of riparian zone on water temperature showed a negative correlation between canopy coverage and water temperature (Broadmeadow *et al.* 2011). Increase of water temperature in restored streams exposed to higher solar irradiance due to absence shadow of canopy has been highlighted, and differences can raise 5-10°C (Daniluk *et al.* 2013). However, this is only a limited number of cases showing the increase of water temperature in the context of river restoration, due to the current lack of scientific interest in this topic and the difficulties to measure precisely water temperature in rivers.

3.7. Increased Spreading Rates of Invasive Species

By Amael Paillex, Eawag, CH

Problem description

Alien species are species occurring outside of the range they occupy naturally or inhabit area they could not occupy without direct or indirect introduction by humans (Mack *et al.* 2000; Daisie 2009). The majority of alien species cause no harm, while some alien species spread very rapidly and can harm biological diversity, human health, and economic values and are considered as invasive (Mack *et al.* 2000). Those species create a dilemma in the case of restoration programmes and a problem for conservation effort (Chapin *et al.* 2000; Strayer 2010). Indeed, river and floodplain restoration aims at recovering dynamic and healthy ecosystems with a reduction of stressors on the ecosystem. However, restored habitats might be an opportunity for alien species to settle in newly created environments and rapidly colonize empty niches. The longitudinal and lateral connectivity increase by restoration works might be an unfortunate opportunity for the establishment of alien species (Strayer 2010), and the change in biotic conditions might favour the establishment of alien species after restoration efforts. Indeed, communities with a high richness are unlikely to have vacant niches available for alien species (Mack *et al.* 2000), while impoverished communities by restoration efforts might be not resistant to invasion. However, this idea is controversial and recent researches suggest that the high richness of a site might not be a barrier against invasion (Stohlgren, Barnett & Kartesz 2003).

Counter-measures

Alien species might have specific characteristics that help them establishing populations in disturbed environments (Sakai *et al.* 2001). Compared to native species, alien species have different functional characteristics (Statzner, Bonada & Doledec 2008), and these characteristics are supposed to enable them to develop dense populations (Sakai *et al.* 2001). Assessing the invasiveness of an ecosystem under a certain level of restoration would be highly desirable to optimize sustainable restoration strategies, because restoration practitioners have no tools to avoid the spread of invasive species in newly restored rivers. Only strategies exist to reduce the impact of introduced invasive species, without possibility to eradicate the species from the sites. The potential introduction of invasive species in newly restored sites should be assessed to provide sound restoration recommendations aiming at the limitation of alien species dispersal and an optimal conservation of biodiversity (Palmer *et al.* 2005).

Case studies

Currently few studies showed the impact of river and floodplain restoration on alien species and functions. On the Rhône River, studies reported the increase of macroinvertebrate alien richness in sites newly restored (Paillex *et al.* 2009; Besacier-Monbertrand, Paillex & Castella 2010). On the Danube River, studies related the increase of alien frequencies in restored sites (Funk *et al.* 2009). On the Rhine, a high amount of non-native species settled in a secondary man-made channel after its connection to the main river channel. In this case, the non-native species came from the main river channel that already comprised 15% of non-native taxa (Simons *et al.* 2001). However, none of these studies disentangle the natural invasion of the sites with new invasive arrivals and the real effect of restoration on encouraging the spread of invasive species. A recent predictive model permitted, however, to show that restoration encouraged alien species more than what could be naturally expected by dispersal of invasive species (Paillex *et al.* 2015). Nowadays, we ignore the reasons why such restoration operations encouraged the alien species, a controversial topic that is relevant to test for a better understanding of biological invasions and management of freshwater ecosystems.

3.8. Riparian Vegetation Encroachment

By Diego García de Jalón, UPM, ES

Problem description

Riparian vegetation interacts with flowing water and sediment acting as a natural control on river morphodynamics (Gurnell *et al.*, 2012; Gurnell 2013a). As a living factor, vegetation distribution in a river changes over time. Its dynamics is the result of a balance between vegetation recruitment and growth against Vegetation uprooting and mortality. Vegetation expands its cover depending on the ability of vegetation to colonize the disturbed environments of river channels and floodplains through sexual (seeds) or asexual (vegetative) recruitment processes. On the contrary vegetation reduces its cover when fluvial processes remove established vegetation to widen or deepen the channel. Vegetation

is removed by high flows that exceed the resistance of plants and uproot them directly, or erodes sediments undermining their root system until they are no longer firmly anchored.

When flow disturbances are reduced and/or vegetative environment (nutrients, light and soil moisture) is enhanced the fluvial balance promotes *vegetation encroachment*. Then vegetation recruitment and growth become dominant, trapping and stabilizing fluvial sediments that favour vegetation advances into the active river channel, producing channel narrowing and bed aggradation around the encroaching vegetation.

Vegetation encroachment into the active channel is one of the most widespread and potentially intractable effects of flow regulation on alluvial river systems (Scott et al. 1996, Stella et al. 2003). In natural alluvial river systems, geomorphic processes such as flooding, erosion, and sediment deposition maintain an equilibrium channel shape and cross section width. Through these processes, the river constructs and maintains a multi-staged channel and a floodplain. Downstream of large dams, flows with reduced magnitude are released, and consequently scour of alluvial bars in the active channel is reduced, allowing riparian trees to become established in the former (predam) active channel.

In semiarid and Mediterranean rivers the composition of encroached riparian systems is changed with an increase of upland non-riparian species (Santos, 2010; Huxman et al. 2005).

Counter-measures

Preventing riparian vegetation encroachment by directly avoiding new recruitment, or eliminating saplings with controlling flows may be a simple task. However, once the vegetation has been established and grown, especially if it is in dense spots, the flushing flows or peaks flows needed to uproot the plants are often unattainable.

Biological control of vegetation encroachment may be an option but it has also secondary effects. Increasing grazing by ungulates may effectively limit the vegetation recruitment and prevent height growth.

Case studies

Marston et al. (1995) describe how Human pressures caused vegetation encroachment process in the R. Ain. The effect of reservoir construction, artificial cut-offs and lateral embankments has been to trigger a shift from a braided to single-thread meandering channel and to accelerate entrenchment. With entrenchment came less disturbance of the floodplain by floods, sedimentation and lateral channel shifts. In addition, the water table dropped in elevation. As a result, the development of floodplain vegetation was altered from a pulse-like disturbance regime to a more terrestrial-like pattern of vegetation succession. Terrestrial-like refers to succession in the absence of channel disturbance. Alluvial forests and very dry grasslands have expanded at the expense of pioneer, disturbance-dependent communities. A positive feedback continues to exist because entrenchment leads to lower disturbance and vegetation encroachment which, in turn, further restricts channel migration and overbank flows.

Gordon & Meemtemeyer (2006) found that operation of the dam and land use patterns together influenced spatial and temporal changes in channel morphology and riparian vegetation in an agricultural stream system in northern California. The area of riparian vegetation was increased 72% after operation of the dam. Across time periods, decreases in the area of riparian vegetation were also associated with increases in land use area in both the dammed and reference stream. After operation of the dam, reduced peak discharges and sediment reduction likely lead to channel incision and constrained channel migration, which allowed vegetation to increase 50% on less accessible, abandoned banks.

Stella et al. 2003 assessed the effects of flow regulation on channel width in the Merced River (California) detecting that vegetation has encroached onto formerly active bars, causing the river channel to become narrower and eliminating the multi-staged form of the channel.

3.9. Increasing Habitats for Mosquitos

By Piet Verdonschot, Alterra, NL

Problem description

One of the main challenges to adapting to climate change has involved the development of national and regional strategies to mitigate inland and coastal flooding through the rehabilitation of former and the creation of new wetlands (Medlock & Vaux 2011). In addition, in an attempt to increase the available habitat for wildlife also wetlands are being created and failing wetlands are being restored through careful wetland management. Many of these strategies involve recreating wetland landscapes in areas where wetlands had previously dominated. These wetlands have been formerly drained for agriculture, urban and other purposes, even the management of mosquito nuisance (Schäfer et al. 2004). In urban areas, new wetlands are being created as part of ecological mitigation for new housing developments, with a strong driver to provide mitigation habitat for European protected species.

Mosquitoes are acutely responsive to changes in temperature and rainfall. The rate of development of is directly proportionate to temperature, as this governs the rate of immature development, blood digestion, and egg production, as well as incidental issues such as pathogen development within the mosquito. However, as an obligate aquatic insect, the degree of water availability has profound implications for the survival and abundance of mosquitoes (Medlock & Vaux 2015).

The water permanence strongly impacts on the mosquito's competitors and predators. Mosquitoes have low competitive abilities and thus often only survive in masses when water levels strongly fluctuate, or waters are intermittent or temporary. Hence, all weather-driven or human-driven processes that affect water permanence (stability), e.g. heavy rainfall or wetland management, will impact to varying degrees on mosquito diversity and density (Bisanzio et al. 2011, Stanke et al. 2013).

Mosquito-borne disease is increasing in incidence in Europe, partly due to the incursion of a number of invasive species in relation to e.g. dengue and chikungunya viruses but also due

to the involvement of native species in transmission of West Nile virus and malaria (ECD 2010, 2011, 2013). For some of these infections, this is a re-emergence, with recognition that mosquito-borne disease was once widespread in Europe, and may become again. Some mosquito species exploit container habitats in urban areas; a product of human-made habitats resulting from increased urbanisation. However many species exist in natural ecosystems in wetlands.

In river valleys, rehabilitation and creation of wetlands act as a storage facility for floodwater. This directly means the creation of temporary water bodies. It is an important question what impact re-installing wetlands in river valleys means for the potential distribution of mosquito-borne diseases and in its turn what that means for the water and vegetation management in such new wetlands to prevent spread of diseases and nuisance.

Counter-measures

Thinking about counter measures asks for a careful consideration of the general life-histories of mosquitoes and how these relate to wetland types (e.g. Cranston et al. 1987). Some mosquito species, often species of the genus *Anopheles*, exploit only small sized permanent wetlands such as ponds and ditches. Small but open and large water bodies tend to be unsuited for mosquitoes as they are subject to competition and predation, and surface movements that prevent immature mosquito survival. Running waters in general are also unsuited due to water movement. The same accounts for water bodies with extensive drawdown zones, such as reed beds. Calm, partly isolated vegetated zone and the presence of floating or emergent vegetation is needed to support mosquitoes in permanent water bodies (Dale & Knight 2008).

A second group of mosquitoes, more often species of the genera *Aedes* and *Ochlerotatus*, thrive in temporary water that is subjected to seasonal flooding and drying. Representatives of wet woodland tend to exploit winter flooded habitats, with immature development occurring during late winter and early spring prior to summer drying. Representatives of wet grassland remain dormant during winter as eggs, awaiting summer floods, upon which immatures develop in late spring for a summer emergence of adults (Schäfer et al. 2004). Ephemeral wetlands that irregularly, routinely dry and re-wet tend to be inhabited by a third group of mosquitoes, mostly from the genera *Culex* and *Culiseta* (Carver et al. 2010).

The main link between mosquito nuisance and wetlands is the hydrological regime. Extreme events such as drought which results in the unnatural drying of permanent wetlands, followed by a re-wetting induces dramatic increases in mosquito numbers in the absence of competitors and predators. It shall be clear that maintaining a more or less permanent water level in wetlands throughout the year is crucial to minimise the occurrence of nuisance mosquitoes (Chase & Knight 2003).

Potential measures are:

- maintaining high water levels in early spring, followed by drawdown in late spring (Malan et al. 2009)
- increasing the rate of water flow (subsurface flow, wind-assisted water movement or human-assisted turbulence (i.e. pumping)) (Dale & Knight 2008)
- improving water quality (reducing nutrient and organic loads) (USDA 2008)
- stimulating predation by e.g. deepening or connecting wetlands (Teels 2009, Chase &

Shulman 2009)

- regular maintenance (removing silt -up) and promoting rapid dewatering (i.e. increased out-flow of water) and preventing pooling (Russell 2009)

Case studies

The most recent and effective approach in mosquito nuisance control is source reduction, which simply means reducing the source of mosquito populations. This does not necessarily mean destroying the wetland, as relatively small changes (often in hydrology) may adversely impact the mosquito life cycle and prevent adult emergence (Dale & Knight 2008). A review of the effects of impoundment is given by Patterson (2004). An extensive overview of the ecological setting and context for mosquito management and control in wetlands, the mosquito abatement options available for current wetlands managers and mosquito control professionals, and an outline of the necessary considerations when devising mosquito management strategies are provided by Rey et al. (2012). Although the cases emphasize the North American wetlands, most of the material is applicable to wetlands elsewhere. Design and maintenance are important for managing mosquito larvae to inhibit adult emergence in newly created and constructed wetlands (Walton & Workman 1998, Mayhew et al. 2004). A growing body of technical information is also available on the ecology and management of mosquitoes in wetlands treating municipal wastewaters (Knight et al. 2003). Metzger (2004) provides basic guidelines for managing mosquitoes in stormwater treatment devices.

3.10. Increasing Accident Rates of Visitors

By Michelle Smith and Ian Cowx, UHULL, UK

Problem description

The environment surrounding rivers, especially in urban areas are often appreciated for relaxation and recreation (Aberg and Tapsell, 2013). River rehabilitation projects have the potential to increase the social value of the river environment by providing attractive places for social activities (Coley, Kuo and Sullivan, 1997 in Aberg and Tapsell, 2013), visits and recreational activities, which can lead to positive emotions and feelings of well-being (Chiesura, 2004 in Aberg and Tapsell, 2013).

Improvements to the aesthetic appearance of the riverscape following river rehabilitation could potentially lead to an increase in the number of people using the river and the surrounding area for recreational activities. For example, following rehabilitation of the River Skerne in Darlington, UK, surveys found a large increase in the percentage of respondents who said they visited the river more frequently following river rehabilitation (Aberg and Tapsell, 2013).

Despite river rehabilitation providing numerous potential social and recreational benefits, there is a need to recognise that increased use of the river and the surrounding environment requires issues such as public health and safety concerns to be addressed. Risks around rivers and the river corridor are often associated with accidental drowning from falling into rivers, canals and lakes whilst walking around (ROSPA). In addition, there is a greater

likelihood of people injuring themselves whilst walking or playing in the surrounding area and along the associated river banks. Increased use of the river and river corridor could therefore potentially cause an increase in the number of accidental drowning incidents or injuries such as twisted ankles.

There are also potential issues associated with public health. Water is often associated with transmission of diseases (<http://www.britishrowing.org/sites/default/files/rowsafe/5-3-WaterborneDiseases-v1.pdf>) such as Leptospirosis, gastro-intestinal illnesses like Cryptosporidium or Hepatitis B. In addition, people walking through scrub and long grasses in the rehabilitated area may be more prone to being bitten by ticks or other insects that carry diseases like Lyme disease (<http://www.lymediseaseaction.org.uk/about-ticks>), which are potentially fatal. Most of the public are not aware of these problems and may be susceptible to increased frequency of contracting the diseases. Issues of public health may also be connected to the volume of litter, either that transported by the river itself or left there by humans, present at a site (Schanze et al., 2004).

Counter-measures

There are two potential approaches to incorporating public health and safety into river rehabilitation. Either, use preventative measures targeting the cause of the health and safety problem, for example lowering steep banks to decrease the risk of the public falling into river and drowning (Schanze et al., 2004). Or, use corrective measures to target the mitigation and management of the existing risk for public health and safety (Schanze et al., 2004). Both approaches can be combined to effectively manage public health and safety during and following river rehabilitation (Schanze et al., 2004). In addition, display boards can be placed at strategic locations such as the entrance to the path or park warning people of the dangers of walking near water and symptoms of the more common diseases.

Case studies

Schanze et al. (2004) indicated that around 30% of the case studies had implemented techniques for accident prevention. Several measures were reported in these case studies including safe bank design, reducing the slope of banks, terracing banks, supplying escape ladders, installing railings, installing bollards to separate pathways and the water's edge and installing fences or trash screens at culverted sections (Schanze et al., 2004). Providing the public with information and education about the risks of flowing water can support physical techniques for accident prevention (Schanze et al., 2004).

4. Uncertainty in CEA/CBA

By Roy Brouwer, IVM-VU Amsterdam, NL

4.1. Introduction

The Water Framework Directive (WFD), adopted in 2000 (2000/60/EC), is one of the first European directives in the domain of water, where economics is an integral part of the decision-making processes surrounding its implementation in Member States. The implementation process has been an important learning process so far for both experts and policy makers. Many decision issues have been clarified in the past years, but many policy and research questions related to the development of the integrated river basin management plans remain open. Besides imperfect knowledge and information about the functioning of the water environment (*natural uncertainty*) and the current and future social and economic values of water systems to human beings (*social uncertainty*), *political uncertainty* exists about the decision issues, the decision criteria, the decision-making procedures and the political, institutional, social, financial and economic consequences of the decisions to be taken. The distinction between natural and social uncertainty is based on Bishop (1979). Bishop distinguishes between natural and social uncertainty, where natural uncertainty derives from imperfect knowledge of the natural environment and social uncertainty concerns variables such as future incomes and technologies that will influence whether or not a resource is regarded as valuable. In the context of political decision-making, a distinction has furthermore been made between goal uncertainty (policy objectives), action uncertainty (composition of alternative sets of measures) and yield uncertainty (costs and benefits of alternative sets of measures) (van Asselt, 2000). A strict distinction between these three sources of uncertainty is questionable and it has been argued that scientific knowledge and the characterization of uncertainty is not independent of political context and are in fact co-produced by scientists and the society and institutions within which they are embedded (e.g. Funtowicz and Ravetz, 1992; Jasanoff, 1996; Sarewitz, 2004).

Uncertainty and its integration in decision-making surrounding the implementation of the WFD was identified as one of the main issues that require further investigation according to the Guidance Document on the Economic Analysis published in 2002 by the Water and Economics Working Group (WATECO) under the WFD Common Implementation Strategy. Following mainstream thinking (e.g. Faber and Proops, 1990; Costanza, 1994; Shackley and Wynne, 1996; Sarewitz, 2004), uncertainty can be generally defined as limited (incomplete or imperfect) knowledge or information about current or future environmental, social, economic, technological, political or institutional conditions, states or outcomes and the implications or consequences of these current or future conditions, states or outcomes. Walker et al. (2003) furthermore distinguish between epistemic and stochastic uncertainty, where the former refers to uncertainty due to imperfect knowledge and information and the latter to uncertainty due to the inherent variability in natural events or phenomena. Assigning probabilities to (reaching) these uncertain (potential) states or outcomes has been the most common strategy to deal with uncertainty in science, policy and decision-making, and provides the basis for the distinction between uncertainty and risk. Here, risk is defined

as the probability of facing or reaching a specific (potential) state, condition or outcome. Awareness of (existing knowledge or information about) potential states or outcomes is a prerequisite in order to be able to assign any probabilities. This is usually referred to as ignorance in the literature (e.g. Faber and Proops, 1990), where ignorance is defined as the lack of awareness of (potential) states, conditions or outcomes. In the EU funded project HarmoniRiB, Brown (2004) distinguished between statistical uncertainty (all outcomes and probabilities known), scenario uncertainty (some outcomes known, no probabilities known), qualitative uncertainty (some outcomes and some probabilities known) and recognised ignorance (no outcomes known).

4.2. Economic Theory Underlying Risk and Uncertainty

In the economic literature, a long-standing debate exists around the distinction between risk and uncertainty. Bernoulli (1738) was the first to introduce the notion of expected utility in the context of risk, decomposing the valuation of a risky venture into the sum of utilities from outcomes weighted by the probabilities of these outcomes. However, it was not until 1921 when the notions of risk and uncertainty were defined separately by Knight in his book 'Risk, Uncertainty and Profit' and not until 1944 when the notion of risk was formalised by Von Neumann and Morgenstern in expected utility theory.

In Knight's interpretation, risk refers to situations where the decision-maker can assign mathematical probabilities to the randomness, which he is faced with. In contrast, uncertainty refers to situations when this randomness cannot be expressed in terms of specific mathematical probabilities. Following this definition, a number of decision theories under uncertainty were developed in economic theory, of which the work by Von Neumann and Morgenstern on expected utility theory (1944) was the first to trigger subsequent work by Savage (1954), allowing objective and subjective probabilities to be determined jointly in expected utility theory, and Arrow (1953) and Debreu (1959) and their state-preference approach to uncertainty where no mathematical probabilities are assigned at all.

Hence, three basic strands of thought can be distinguished in economic choice theory under uncertainty: one assigning objective probabilities to random events (Von Neumann-Morgenstern), one not assigning any probabilities to random events because of lack of knowledge and arguing that probabilities really are only beliefs with no necessary connection to the true randomness of the world, if random at all (Arrow and Debreu), and one assigning objective and subjective probabilities to random events (Savage). Contrary to the utility function in decision making without uncertainty, which is only able to rank alternatives, the Von Neumann-Morgenstern utility function has the advantage that it is able to also measure the strength of preferences over outcomes (van Zandt, 2003). It is this expected utility theory, measuring the strength of preferences, which underlies neo-classical economic welfare theory today. However, opinions differ regarding the assigning, nature and interpretation of the (un)known probabilities.

Early definitions of risk and uncertainty in the environmental economics literature include for example the ones given by Faber and Proops (1990) and Costanza (1994). According to Faber and Proops (1990), risk means that future events are not known for certain, but can be associated with known objective or subjective probability distributions, while uncertainty

means that some future events are definitely unknowable and cannot be associated with probability distributions based on past knowledge. Costanza (1994) defines risk in a similar way as an event with a known probability and true uncertainty where objective or subjective probabilities are not known.

Most economic choice theories nowadays treat uncertainty as a state that can in principle be known through objective or subjective probability distributions and preferences for specific probability distributions (e.g. risk averse, risk neutral or risk loving attitudes or behaviour). In assuming that all uncertainty can be quantified by given probabilities, the economic approach treats the problem as an analysis of risk rather than true uncertainty (Crowards, 1996). Although this approach is powerful for non-complex and non-dynamic systems, such as the transaction between a household and a producer, it is not considered adequate for integrated economic-biophysical-social systems with new currently unforeseen or unknown (i.e. novel) and uncertain future outcomes (de Wit, 2002). The degree to which a studied future system can be treated as a risk or as an uncertainty, depends to a large extent on the novelty contained in the system. When the system contains no or little novelty, probabilistic approaches are sufficient. However, when the degree of novelty increases, due to complexity and dynamics in the system, probabilistic approaches would not be sufficient to predict and manage future events. According to Costanza (1994, p.97), '*most important environmental problems suffer from true uncertainty, not merely risk*'.

4.3. The Economic Value of Additional Information

The reduction of uncertainty surrounding scientific knowledge and information is the standard model for informing the correct course of action (Sarewitz, 2004). Uncertainty in decision-making can in some instances be reduced by new research and the availability of new information. New information can change (1) the *a-priori* expectations, probabilities or beliefs about certain outcomes (into *posterior* beliefs), (2) the potential outcomes themselves and (3) preferences over acts or actions (van Zandt, 2003). Decision-making conditional on additional information and the corresponding updating of beliefs incorporating the newly acquired information is analyzed using Bayes' theorem. According to the Bayesian approach, an agent's beliefs are represented by a subjective probability measure or Bayesian prior. There is no meaningful distinction using this approach between risk, where probabilities are available to guide choice, and uncertainty, where information is too imprecise to be summarized adequately by probabilities (Miao and Wang, 2004).

Alternatively, decision-makers can wait to take a decision based on expectations about future changes. For example, farmers who carry large losses may be rationally keeping their operation alive on the chance that the future will be brighter (Dixit, 1992, p.109). In other words, waiting has positive value. Agents decide when to exercise an 'option' analogous to a financial call option – a decision-maker has the right but not the obligation to buy an asset at some future time of its choosing. This options approach has been widely applied in investments and corporate finance (Dixit and Pindyck, 1994). In the environmental economics literature (e.g. Pearce and Turner, 1990), the concept of option value has been introduced, which is akin to the insurance premium that a risk-averse individual is willing to pay to maintain a resource for future use. However, this risk premium, as an addition to expected consumer surplus, still relies on assigning probabilities to a number of alternative

outcomes, each with known payoffs (Crowards, 1996). Quasi-option value refers to the gains from delaying a decision that would impinge on the environment, contingent on acquiring improved information in the future. This too is founded on expected utility theory, based on (subjective) probabilities assigned to future, known outcomes. Therefore, whilst these option values may help to estimate the full benefits of preservation, they do not solve the problem of true uncertainty in the context of irreversibility (Bishop, 1978; Crowards, 1996).

4.4. Dealing with Uncertainty in Practice

Except for the use of probabilistic (Bayesian) approaches, there exist no detailed, practical, prescriptive guidelines on how to conduct an uncertainty analysis in economic analyses. Box 4.1 below presents, as an illustration, the guiding principles to uncertainty assessment in economic analysis provided by the US Environmental Protection Agency.

The situation is more or less the same for the practical implementation of environmental policy in Europe and elsewhere. General guiding principles are advocated, but no standard prescription is available. Examples are the precautionary principle or the adoption of safe minimum standards. Costanza and Cornwell (1992) argue that one of the main reasons for the problems with current methods of environmental management tackling uncertainty are not just related to its existence, but the radically different expectations and modes of operation that scientists and policymakers have developed to deal with it. The notions of the precautionary principle or safe minimum standards when facing true uncertainty (for example in the context of (ir)reversible climate change) suggest clear decision rules to avoid environmental degradation beyond certain threshold values. However, there is a distinct lack of clarity surrounding the science, economics and politics of what such values entail (Crowards, 1996).

Box 4.1: Guiding Principles for Uncertainty Analysis

Uncertainty is inherent in economic analyses, particularly those associated with environmental benefits for which there are no existing markets. The issue for the analyst is not how to avoid uncertainty, but how to account for it and present useful conclusions to those making policy decisions. Treatment of uncertainty, therefore, should be considered part of the communication process between analysts and policy makers. Transparency and clarity of presentation are the guiding principles for assessing and describing uncertainty in economic analyses. Although the extent to which uncertainty is treated and presented will vary according to the specific needs of the economic analysis, some general minimum requirements apply to most economic analyses. In assessing and presenting uncertainty the analyst should, if feasible:

- present outcomes or conclusions based on expected or most plausible values;
- provide descriptions of all known key assumptions, biases, and omissions;
- perform sensitivity analysis on key assumptions; and
- justify the assumptions used in the sensitivity analysis.

The outcome of the initial assessment of uncertainty may be sufficient to support the policy decisions. If, however, the implications of uncertainty are not adequately captured in the initial assessment then a more sophisticated analysis should be undertaken. The need for additional analysis should be clearly stated, along with a description of the other methods used for assessing uncertainty. These methods include decision trees, Delphi-type methods, and meta-analysis. Probabilistic methods, including Monte Carlo analysis, can be particularly useful because they explicitly characterize analytical uncertainty and variability. However, these methods can be difficult to implement, often requiring more data than are available to the analyst. Confidence intervals are generally useful to describe the uncertainty associated with particular variables. When data are available to estimate confidence intervals they can serve to characterize the precision of estimates and to bound the values used in sensitivity analysis.

Source: Guidelines for Preparing Economic Analyses, United States Environmental Protection Agency, September 2000.

4.5. Uncertainty in Cost-Effectiveness Analysis

According to the WATECO guidance (WATECO 2002), the definition of the program of measures and the ranking of possible basic and supplementary measures based on cost-effectiveness criteria is the key economic input into the preparation of the river basin management plans. The main steps identified in the guidance include the estimation of the costs of each measure, the estimation of the effectiveness (environmental impact) of each

measure and the ranking of cost-effectiveness of measures. Hence, the information needed basically relates to the costs of potential measures and their effectiveness. It is here where most of the natural, social and political uncertainty comes together as the work presented in the previous chapters is an integral part of the cost-effectiveness analysis.

The cost-effectiveness analysis should be conducted at the river basin level. In some cases, it may be more practical to undertake the analysis for sub-basins. However, the hydrological integrity of the basin needs to be kept, starting for example with the most up-stream sub-basin and working downwards. According to WATECO, specific care needs to be given to the choice of the effectiveness indicator as different effectiveness indicators may lead to different rankings of measures. Furthermore, specific attention may be required as the effectiveness of measures can often be assessed qualitatively only for a few environmental indicators, and not for the range of environmental issues encompassed in the definition of good water status. Attention should also be paid to the assessment of different costs in the cost-effectiveness analysis. Often, information may not be available for specific cost types.

Finally, WATECO states that uncertainty about costs, effectiveness and time-lagged effects of measures needs to be dealt with throughout the economic analysis process, and more generally throughout the process of identifying measures and developing the river basin management plan. Sources for uncertainty are highly diverse according to situations and river basins, but will exist with regards to the assessment of pressures, impacts, baseline scenario, costs or effectiveness. It is important that key areas of uncertainty and key assumptions made for the analysis are clearly spelt out and reported along the results of the analysis, so a comparison between analyses undertaken in different river basins and regular updates of the analysis will always be possible. A sensitivity analysis is required for assessing the robustness of the results.

The purpose of a cost-effectiveness analysis is to find out how predetermined targets, e.g. nutrient loads in a catchment or estuary or the degree of naturalness of rivers, can be achieved at least costs. Theoretically speaking, the least cost allocation of river restoration strategies is found if and only if the marginal costs of the proposed restoration measures are equal. The marginal costs of these restoration measures can for example be defined as the increase in total costs when restoring one additional meter of river or stream. As long as marginal costs of river restoration measures are not equal, it is theoretically possible to obtain the same level of restoration at lower costs by shifting from high cost measures to lower cost measures. In principle, a cost-effectiveness analysis answers the question how and how much river restoration is needed in order to be able to achieve the predetermined policy objectives at least costs. In the WFD a cost-effectiveness analysis has to be carried out at catchment level. Hence, also the spatial distribution of costs plays an important role and the question where the restoration measures should be taken. The various steps distinguished in a cost-effectiveness analysis are described in Box 4.2. These steps are taken in sequence, but important feed-backs may exist between steps as learning more about the problem, the source-effect pathway and possible solutions, the same step may be revisited several times. The outline of the various steps shows that carrying out a cost-effectiveness analysis is a multi-disciplinary exercise, requiring the input and collaboration of different scientific disciplines, such as natural scientists and economists, and technical

engineers, but also the input and collaboration of policy and decision-makers as they determine the scope and objective of the analysis.

Box 4.2: Steps in a cost-effectiveness analysis

Step 1: Identify the environmental objective(s) involved

Step 2: Determine the extent to which the environmental objective(s) is (are) met

Step 3: Identify pressures and impacts now and in the future over the appropriate time horizon

Step 4: Identify restoration measures to bridge the gap between the reference (baseline) and target situation (environmental objective(s))

Step 5: Assess the effectiveness of these measures in reaching the environmental objective(s)

Step 6: Assess the costs of these river restoration measures

Step 7: Rank the river restoration measures in terms of increasing unit costs

Step 8: Determine the least cost way to reach the environmental objective(s) based on the ranking of the restoration measures

Following the terminology introduced in the Introduction of this chapter, the main uncertainties underlying the cost-effectiveness analysis of proposed programmes of measures include:

- 1) Political and natural scientific uncertainty about the environmental goals and parameters: i.e. the target situation and how the achievement of this target situation is measured. In view of the fact that both the outcome (good ecological status) and probability of reaching the outcome are uncertain, this mainly involves qualitative uncertainty.
- 2) Natural uncertainty about the pressures involved and the extent to which these pressures contribute to (impact on) water quality problems through the often complex environmental source-effect chain in time and space. This includes what has been referred to as scenario uncertainty by Brown (2004) about possible future pressures.
- 3) Natural uncertainty about the effectiveness of proposed river restoration measures on the ecology of the water system (again in time and space). This largely concerns what Brown (2004) calls qualitative uncertainty, but sometimes also recognized ignorance.

- 4) Social uncertainty about the direct and indirect costs of proposed measures. Indirect costs are related to the fact that an economic activity has various forward and backward links to other economic activities. Interventions in one activity may therefore result in a chain reaction throughout the entire economic system (depending on the nature and the extent of the intervention, e.g. involving large scale land use changes). In order to be able to assess these multiplier effects accurately, a profound understanding of the structure of the economy at river basin scale is required. Given current understanding of these economic relationships at regional levels, statistical and some degree of qualitative uncertainty plays a role here.

Conclusions

This section presented a discussion about the natural, social and political uncertainties surrounding the selection of a cost-effective programme of measures. According to WATECO the cost-effectiveness analysis is the key economic input into the preparation of the river basin management plan and uncertainty about costs, effectiveness and time-lagged effects of measures need to be dealt with throughout the process of identifying measures, including river restoration, and developing the basin management plan. The main objective of this chapter was to demonstrate that uncertainty plays a role in each single step of the cost-effectiveness analysis. The identification of environmental objectives is the first essential step in the cost-effectiveness analysis. The uncertainties surrounding the process of objective identification are partly based on natural uncertainties regarding the most appropriate reference state of 'good water status' (some undisturbed pristine state 100, 200, 500 or 1000 years ago?) and the feasibility of reaching such a state in practice (qualitative uncertainty) using river restoration, and partly on socio-political uncertainties regarding the current and future social value of good water status and the socio-economic, legal, institutional and administrative implications and consequences of this new state (also largely qualitative uncertainty). Moreover, often no clear-cut scientific link can be established between a river's baseline ecology and its degree of naturalness.

The second step (problem definition/gap analysis) is mainly surrounded by statistical uncertainty about the current natural status of water systems based upon available monitoring data (or the lack thereof). Some degree of more qualitative uncertainty may also play a role as a result of latent or time lagged effects of pressures from the past (e.g. hydrological or hydraulic infrastructure modifications) on current and future water status. The future status of water systems is closely related to the third step where scenario uncertainty about future pressures and impacts on water status have to be determined. This includes natural, social and political uncertainty.

The identification of possible measures to bridge the gap between the expected and desired water status in step 4 may involve some degree of socio-political uncertainty with respect to the social and political acceptability of certain measures, such as large scale agricultural land use changes. Uncertainty about this latter consideration (the scale of the intervention) may be a dominant factor of influence in the final selection of the programme of measures, catered for in the WFD in Article 4 (labelled 'disproportionate costs'), and should theoretically follow the estimation of the most cost-effective programme of measures in step

8, but will in practice often also play a role during the selection of possible measures already.

The fifth step (assessment of the effectiveness of measures) is surrounded by natural uncertainty given the often weak scientific underpinning of dose-effect relationships surrounding river restoration on river ecology and biology. The same applies to step 6 where the costs of the possible measures are estimated. Also here social uncertainty about the direct and indirect financial and economic effects across different groups in society may play a role. The indirect effects of the possible measures depend on the nature and the scale of the foreseen implementation of these measures in different sectors. In the WFD indirect effects are expected to play an important role in for example the agribusiness (primary agricultural sector and associated businesses such as food and beverages processing industry). However, indirect effects as a result of changes in household (consumer) behaviour (consumption patterns) may also play a role if a large share of the implementation costs are passed on to households (directly through prices and taxes for water services or indirectly through an increase of other product prices). Hence, to what extent and through which mechanisms indirect effects manifest themselves is uncertain.

The various steps in the cost-effectiveness analysis are so interdependent and interrelated that it is hard to indicate at which stage (or step) most uncertainty occurs. Uncertainties most likely accumulate when going through the various steps. However, natural science uncertainties surrounding existing cause-effect relationships in water systems, including uncertainties about the spatial and temporal dimensions of river restoration, seem the predominant source of uncertainty in the cost-effectiveness analysis, manifesting itself in the first steps of the identification of environmental objectives and problem definition (degree of naturalness) as a result of the qualitative uncertainty and partly recognized ignorance surrounding cause-effect relationships and in subsequent steps as a result of the same uncertainties surrounding environmental dose-effect relationships of possible restoration measures.

4.6. Uncertainty Surrounding the Economic Valuation of Ecosystem Services Provided by River Restoration

Based on the meta-analysis presented in REFORM Deliverable D5.2 (Brouwer et al. 2015), we discuss here the uncertainties surrounding the benefits assessment of river restoration projects. We do this by further analysing the results from the meta-analysis and zooming in on the mean value estimates found in the literature and the reliability of the estimated meta-regression model for the purpose of benefits transfer. Benefit transfer is the use of existing benefit estimates in the literature to predict the value of the benefits of new river restoration projects. It has been argued in the literature (e.g. Brouwer, 2000) that the use of value functions is to be preferred over and above the use of mean value estimates. More specifically we examine the change in prediction error when using the estimated meta-regression model instead of mean unit values. We briefly summarize the main results from the meta-analysis first and then investigate the prediction errors.

4.6.1. Meta-analysis results

The distribution of mean WTP estimates across the entire database is skewed, with the mean value being € 69.9 per household per year and the median € 43.1 per household per year (see Figure 4.1). Although there are some differences in WTP estimates averaged across world regions, e.g. € 66.5 for Europe, € 64.0 for Asia, and € 76.9 for America, statistical tests does not indicate that there are significant differences between these values. The Kruskal-Wallis test for the equality of the mean WTP distribution across different regions reports a p-value of 0.60. At the same time, as Figure 4.2 shows, there is much more variation at individual country level, with mean WTP ranging from € 11.3 for Korea to € 118.0 for Scotland.

Comparing the mean WTP values across different elicitation methods, we find that the average WTP value derived from choice experiments (€ 95.5) is significantly higher than the average WTP value for contingent valuation studies (€ 52.3), with a p-value for the associated Kruskal-Wallis test statistic equal to 0.008. However, differences in average WTP values for the different CV elicitation formats are contrary to expectations not statistically significant.

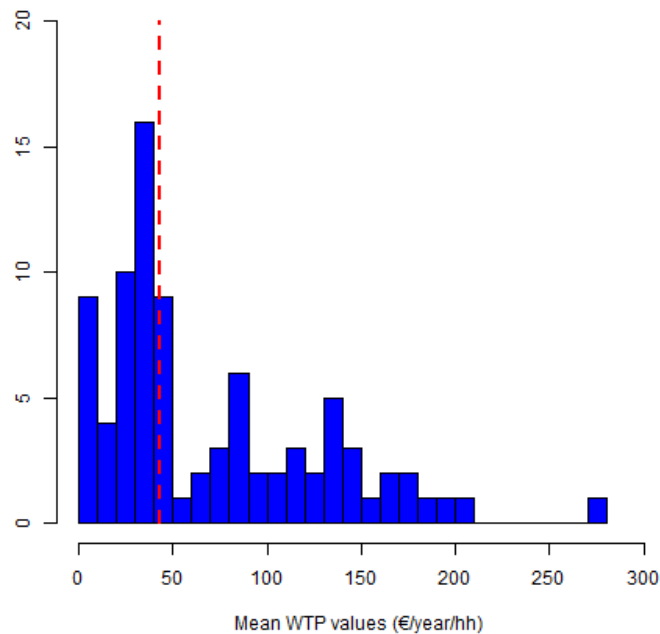


Figure 4.1. Histogram of mean WTP values for river restoration across all regions, in 2013 euro prices per household per year (red line indicates the median WTP estimate)

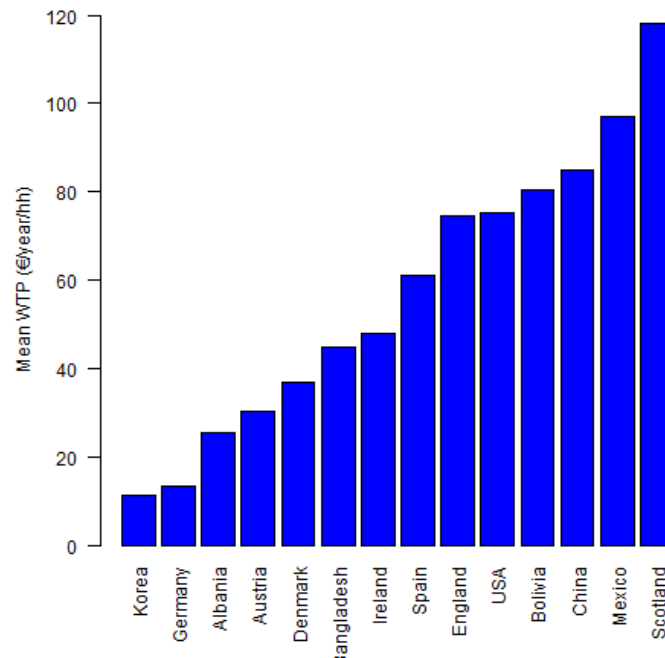


Figure 4.2. Ordered mean WTP distribution across countries, in 2013 euro prices per household per year

A mixed-effects multivariate regression panel model was estimated to test the influence of covariates simultaneously and address both between-study and estimate heterogeneity (see D5.2). For the multivariate meta-analysis we used 29 studies with 107 individual data entries (WTP estimates) in the database. In the process of model selection, several models were estimated that include the main characteristics of the river restoration project, the ecosystem services involved in the valuation scenarios, and the socio-demographic characteristics of the respondents. Categorical variables are coded as dummies, and the continuous variables, such as estimated WTP, average household income, population density, and fraction of the river length studied in a particular river restoration project, are transformed into their natural log form to improve the model fit, and allow for easy interpretation of the coefficient estimates.

The estimation results for the statistically best-fit model, which includes characteristics of the river and ecosystem services, site and population characteristics, as well as characteristics of the valuation method, are presented in the first column of Table 4.1. The overall fit of the model is very good, and the fixed effects explain 68 per cent of the observed variance. Compared to provisioning services such as drinking water and irrigation water supply (the baseline category in the estimated models), WTP for the regulating service flood control is significantly lower and WTP for the regulating services water quality control and erosion control significantly higher. All else being constant (*ceteris paribus*), mean WTP for the cultural services river recreation and landscape aesthetics (role of restored rivers in landscape valuation) is significantly higher compared to provisioning services.

Only in the reduced model do we find a significant positive effect for the fraction of the river that is being restored. However, once we include control for the ecosystem services, this effect becomes insignificant. EU respondents have a significantly lower WTP than respondents elsewhere in the world (US, Asia, Latin America). Also, WTP in more densely populated areas is, as expected due to higher overall demand, significantly higher. Higher income results as expected in a significantly higher mean WTP in the full model. Unfortunately no significant differences are found between users and nonusers.

With respect to the methodological study characteristics, discrete choice experiments generate significantly higher WTP values than CV studies, all else being constant. No significant differences exist between face-to-face (the baseline category) and web-based surveys. Mail surveys, however, generate significantly higher WTP values for river restoration than face-to-face interviews. When asked to pay on behalf of someone's entire household, this significantly reduces mean WTP compared to asking for someone's individual WTP (the baseline category). No significant effect of payment frequency can be detected, which is contrary to the temporal embedding effects observed for example by Stevens et al. (1997), Kim and Haab (2003), Spaninks and Hoevenagel (1995), and Brouwer et al. (2008). Also, a significant effect is found for payment vehicle: conform findings in Brouwer et al. (1999) for wetland ecosystem services, taxes reduce WTP significantly compared to other payment vehicles such as fees (e.g. entrance fee).

Table 4.1. Estimated meta-regression models

Variable	Full meta-model	Reduced transfer model (1)	Reduced transfer model (2)
Intercept	-0.798 (2.139)	1.092 (2.018)	0.358 (2.937)
River and location characteristics			
Location (Europe=1)	-0.991** (0.418)		
Restored river fraction (0-1)	-0.173 (0.506)	1.178* (0.606)	0.771 (0.796)
Population density (people/km ²)	0.309*** (0.079)	0.016 (0.102)	0.178 (0.110)
Population characteristics			
River user (dummy)	0.245 (0.278)		
Average income (€/yr)	0.349* (0.199)	0.196 (0.196)	0.085 (0.278)

Valued ecosystem services

Water quality control	1.602*** (0.247)	1.238*** (0.268)
Flood protection	-2.978*** (0.408)	-3.585*** (0.455)
Erosion protection	0.418* (0.238)	0.352 (0.261)
Recreational amenities	0.400** (0.188)	0.287 (0.201)
Landscape aesthetics	0.759*** (0.159)	0.716*** (0.168)
Biodiversity	0.255 (0.195)	0.127 (0.210)

Study characteristicsValuation method

Choice experiment	0.589** (0.299)
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Administration mode

Web-based survey	0.042 (0.509)
Mail survey	1.059*** (0.400)

Payment characteristics

Household (instead of individual)	-1.699** (0.665)
Payment frequency (1 = less than annual)	-0.349 (0.394)

Payment vehicle

Water bill	-0.358 (0.391)
Tax	-1.411*** (0.451)
Income tax	-3.465*** (0.904)

Model summary statistics

Log likelihood	-94.8	-168.5	-112.8
R ² (fixed effect)	0.68	0.09	0.49
R ² (overall)	0.89	0.38	0.95
AIC	233	349	249
Number of observations	107	107	107

Note: *p<0.1; **p<0.05; ***p<0.01

We also estimated the smallest possible reduced meta-regression models, for value function transfer purposes, for which the results are presented in the second and third columns of Table 4.1. If we only include easy measurable variables based on available secondary data sources like the fraction of the river that will be restored, population density and income, only the first variable is significant. This result is interesting: the higher the share of the river restored, the higher WTP (=sensitivity to scope). Although positive, the estimated coefficients for income become insignificant. Also the effect of population density disappears.

There is also a reduced form model that includes the ecosystem services (reduced model 2). Notably, this model shows much better fit compared to the reduced model 1. In this case the same ecosystem services are significant again except recreation and erosion protection. And only population density is marginally significant, as the fraction restored becomes insignificant, and income remains insignificant.

4.6.2. Prediction errors

In this section, we report the transfer errors for the full best-fit model and the two reduced models, and compare these estimates with the transfer errors for the fixed-effect-size model, i.e. when we take the average WTP to be the best predictor for observed WTP estimates, and there is no need to include any control for other explanatory variables. This allows us to conclude how good the models are in terms of predictive power to assist in future benefit transfer exercises and support policy and decision-making.

The transfer errors are calculated as out-of-sample relative prediction errors, where one observation is omitted from the sample, the model is re-estimated, and a new predicted WTP value is calculated. The resampling is done by a jackknife procedure for each meta-analysis model. Table 4.2 reports the average results (mean, median, and standard deviation of transfer errors) that are based on the jackknifed samples, i.e. across all possible one-entry data omissions. The most notable result is that the full regression model reduces the prediction error by an order of magnitude compared to the simple average WTP model, and substantially reduces error variance of the predicted WTP values. The second reduced model that includes the variables for the ecosystem services also performs well compared both to the average WTP and first reduced models.

Hence, including control for fraction of the river that is restored, population density and income reduces the prediction error by almost a factor 3 compared to simply transferring mean WTP values. Adding in control for the ecosystem services further reduces the prediction error by almost a factor 4. The full model yields the lowest prediction error of, on average, 30 percent.

Table 4.2. Transfer errors for different models

	Mean WTP model	Best-Fit full model	Reduced model 1	Reduced model 2
Mean	10.85	0.31	4.02	1.07
Median	0.53	-0.09	-0.16	-0.16
std. dev.	53.88	1.22	20.90	4.92

We also test for differences in sampling distributions of mean transfer errors for different meta-analysis models. Several two-sample tests, such as Wilcoxon and Kruskal-Wallis, deliver mostly comparable results. First, the difference between average transfer errors for the simple FES model and for the full ME model is highly significant (p-value is less than 0.01), indicating that the latter significantly outperforms the former. Similarly, the differences in mean transfer errors for the FES model and for any of the reduced models are significant at 0.01-level. However, the evidence for the differences between the full and reduced models is somewhat mixed, as different tests lead to conflicting conclusions about the significance of differences in mean transfer errors in this case.

In conclusion, the meta-regression model clearly outperforms the use of average unit values when using existing estimates from the literature for the approximation of the benefits in cost-benefit analysis of new river restoration projects.

5. Uncertainty in MCDA

By Peter Reichert, Nele Schuwirth and Amael Paillex, Eawag, CH

5.1. Introduction

Multi-criteria decision analysis (MCDA) has originally been developed as an approach to support rational decision making by individual decision makers (Keeney and Raiffa 1976; Keeney 1982; Keeney 1992; Belton and Stewart 2001; Eisenführ et al. 2010; Clemen and Reilly 2013). However, the advantages of such a structured approach for societal decision making and for transparently negotiating and communicating decisions has been realized in environmental decision support in general (Salminen et al. 1998; Lahdelma et al. 2000; Kiker et al. 2005; Mendoza and Martins 2006; Hajkowitz 2008; Mahmoud et al. 2009; Huang et al. 2011; Gregory et al. 2012; Linkov and Moberg 2012; Reichert et al. 2015a) and, more specifically, in river rehabilitation planning (Reichert et al. 2007; Beechie et al. 2008; Corsair et al. 2009; Convertino et al. 2013, Reichert et al. 2015).

Figure 5.1 provides an overview of a structure underlying a decision making process as it can be supported by MCDA.

Problem definition (Fig. 5.1, step 1) and stakeholder analysis (Fig. 5.1, step 2) are extremely important steps of decision support, as not being broad enough may lead to ignoring relevant alternatives and being too broad leads to an unnecessarily high effort for the analysis and as the relevant stakeholders depend on problem framing. In the current context, a very important decision is whether to do an MCDA that corresponds to a Cost-Effectiveness Analysis (CEA) or to a Cost-Benefit Analysis (CBA). This can best be illustrated with the next step of an MCDA, of developing an objectives hierarchy by hierarchically breaking down the overall fundamental objective to be achieved (Fig. 5.1, step 3). Fig. 5.2 shows an example of such an objectives hierarchy for river management. The hierarchy shown in Fig. 5.2 is the equivalent to a CBA including the quantification of the benefits. This analysis intends to find alternatives with higher benefits than costs. Alternatively, an MCDA could be formulated to support maximizing the ecological gain under a given budget constraint or for finding the cheapest alternative that leads to a prescribed ecological improvement. This would then correspond to a CEA and would only require the branch in Fig. 5.2 that quantifies the good ecological state of the river. No explicit quantification of the benefits would then be needed. This would apply in particular to the Swiss case study as the budget is already accepted and we should just optimize the ecological gain achieved given the budget constraint. Nevertheless, to demonstrate the application in a broader context, we will perform an MCDA that corresponds to a CBA as shown in the objectives hierarchy in Fig. 5.2. The reduction to an MCDA equivalent to a CEA is then obvious. As a next step (Fig. 5.1, step 3 continued), the degree of fulfillment of the objectives as a function of measurable system attributes is elicited from experts for ecological objectives and from stakeholders or a sample of the population for societal objectives and trade-offs. This is done by following the objectives hierarchy from lower to higher levels. At the lowest level, such value functions can be expressed directly as functions of attributes (only a small number of attributes is needed for each sub-objective, as these are relatively narrow at this level). At higher levels, aggregation procedures are elicited to get the value (degree of fulfillment of the objective)

at the higher level as a function of the values at the lower level (and, indirectly, as a function of the attributes). For the ecological branch of the objectives hierarchy, existing ecological assessment procedures for rivers can be translated into value functions (Langhans et al. 2013). The resulting values on a scale from 0 to 1 can be translated into ecological quality classes by dividing this interval into sub-intervals of equal length. For the typical division into five quality classes, this results in sub-intervals of length 0.2.

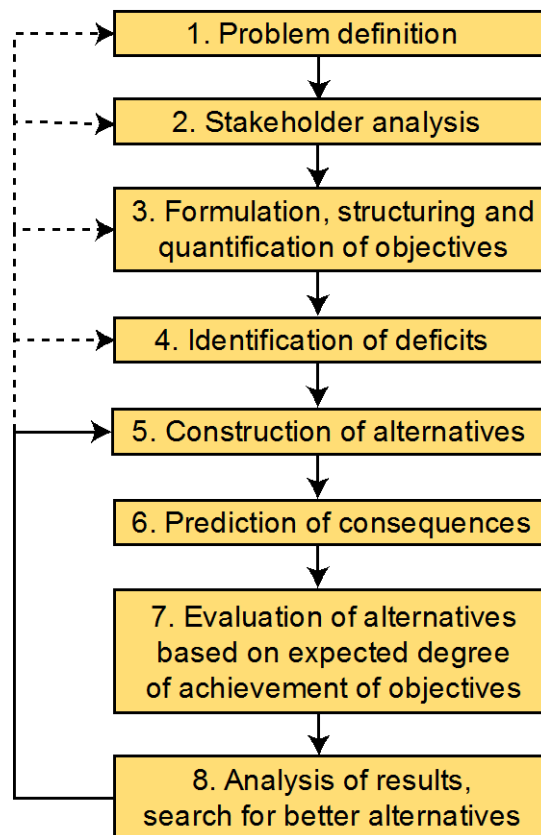


Figure 5.1: Structure of the decision making process according to Reichert et al. (2015) (modified from Schuwirth et al. 2012).

Applying this value function to the current state (represented by observed attributes), leads to the identification of deficits (Fig. 5.1, step 4) that can stimulate the construction of alternatives to improve this state (Fig. 5.1, step 5). Application of the value function representing the preferences to the predicted attributes for all alternatives leads to the valuation for these predicted states and a ranking of the alternatives (Fig. 5.1, step 7). Note that at the “societal level” of the objectives hierarchy, this results in a trade-off between the objectives of a good ecological state and good ecosystems services versus low costs. If an additive value function is used and the weights are proportional to the ranges of monetary costs and benefits, this is exactly equivalent to a CBA, just formulated in “value units” rather than monetary units. In most cases, we will use an additive value function at this level. When doing MCDA, it is possible to include the full ecological assessment procedure into the same framework of a value function (Langhans et al. 2013). In this case, it is important that at the lower hierarchical levels, non-additive aggregation techniques can be used (Langhans

et al. 2014a). This is not possible in a CBA as it does not make sense to non-additively aggregate monetary values. This problem is circumvented in CBA by assessing the ecological state with the existing (not necessarily additive) assessment procedures and only translating its outcome to monetary units for applying the CBA at the higher hierarchical level. Interesting features of MCDA are (i) that an analysis of deficits from the top level to the bottom level can be done within the same framework, (ii) that non-additive aggregation is possible, (iii) that not all values must be expressed in monetary units, and (iv) that risk attitudes can be considered relatively easily, as discussed in section 5.4. As a final step, the results can be analyzed and better alternatives may be constructed (Fig. 5.1, step 8).

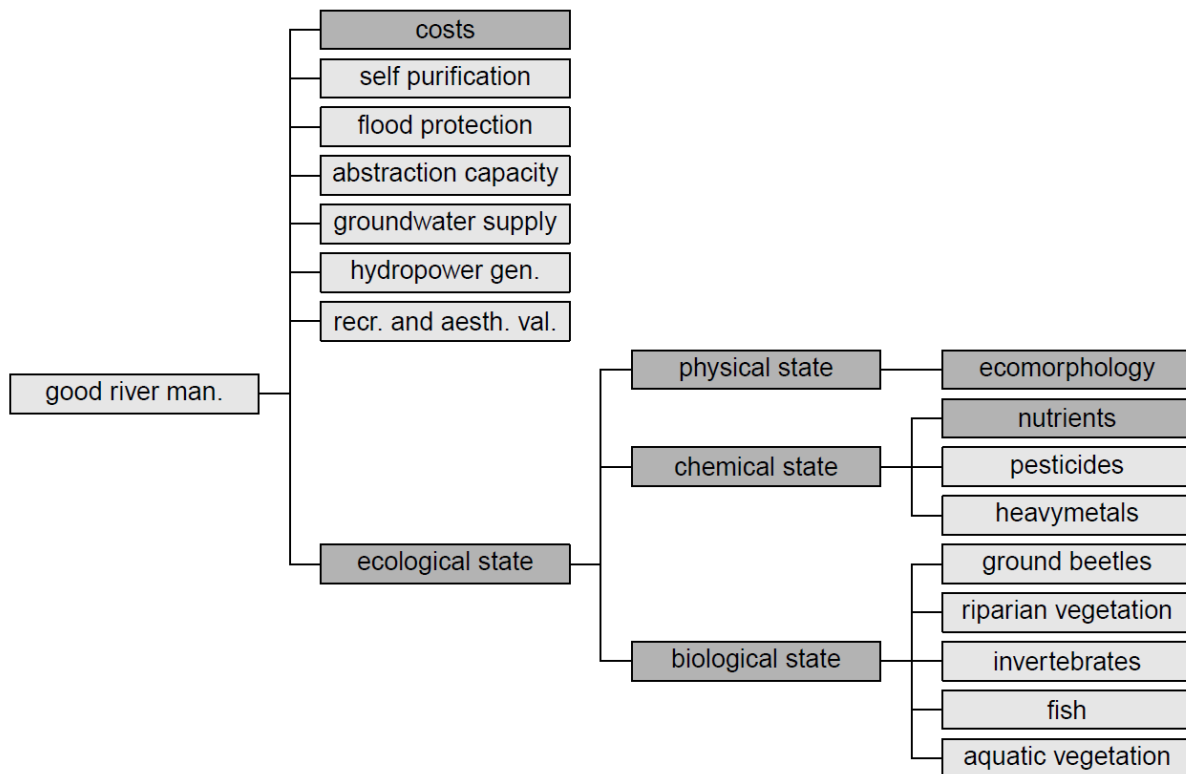


Figure 5.2: Objectives hierarchy of river management. Dark boxes indicate sub-objectives the evaluation of which is mandatory for evaluating the associated sub-objective at the next higher hierarchical level.

In the following sub-sections we will discuss, how to include uncertainty into the MDCA approach outlined in Fig. 5.1 and illustrate this with an application to the same case as already discussed without uncertainty in deliverable D5.2 (Brouwer et al. 2015). In particular, we will focus on uncertainty regarding scientific predictions (Fig. 5.1, step 6), uncertainty in the formulation of preferences including risk attitudes (Fig. 5.1, step 3).

5.2. Uncertainty in Scientific Predictions

As described in chapter 2, uncertainty in scientific predictions are considered by formulating the predictions as probability distributions conditional on the assumptions underlying the decision alternatives, and potentially conditional on future scenarios. These probability distributions should represent intersubjective beliefs of a representative number of scientists knowledgeable about the field (Reichert et al. 2015).

An option of integrating and summarizing uncertain knowledge from diverse sources, such as data, models and experts, is to formulate a probability network model or a Bayesian belief network (McDonald et al. 2015). Such a model consists of a graphical model that sketches the main causal relationships and divides the overall model into submodels, and of a quantification of these submodels as conditional probability distributions. The graphical part is perfectly suited for communication with stakeholders, the quantitative part for compiling all available scientific information. The results have then to be communicated as probability distributions that summarize the scientific knowledge and its uncertainty of the outcomes.

5.3. Uncertainty in the Quantification of Societal Preferences

Uncertainty in societal preferences or their quantification can best be dealt with by sensitivity analysis (Scholten et al. 2014; Scholten et al. 2015). The final ranking of alternatives can then be investigated regarding its sensitivity to preference formulations, e.g. in the form of probability distributions of value functions.

5.4. Consideration of Risk Attitudes

Uncertainty in prediction of attributes can be propagated to values as shown in section 5.2. However, to derive a unique ranking of uncertain alternatives, the risk attitude has to be considered. If a person is risk neutral, he or she evaluates an alternative just based on the expected value of the value function. For example, this risk neutral person would be indifferent between a certain alternative with a value of 0.5 and an alternative with a 50% probability of 0 and 50% probability of 1. If a person is risk averse, he or she prefers a certain alternative to an uncertain alternative with a higher expected value. If a person is risk seeking, he or she prefers an uncertain alternative with a lower expected value (but a chance to get a higher value) than the value of a certain alternative.

To rank uncertain alternatives, the Multi-Attribute Utility Theory (Keeney & Raiffa 1976) can be applied. This theory is grounded in the expected utility theory (Von Neumann and Morgenstern 1944). While (measurable) value functions describe the fulfilment of objectives on an interval scale, where the differences in value show the strength of preference, utility functions combine the strength of preference with the risk attitude. They only provide an ordinal scale. Therefore, the absolute numbers and differences on the utility scale do not have a meaning. They are just used to rank alternatives based on their expected utility.

Elicitation of utility functions: It is possible to elicit utility functions directly as a function of attributes or to convert value functions to utilities at any level of the objectives hierarchy.

Since the elicitation of utility functions is much more demanding than the elicitation of value functions, it is recommendable to elicit value functions first and then convert them to utilities the highest level of the objectives hierarchy (Dyer and Sarin 1982). One method to elicit utility functions is the certainty equivalent method (e.g. Eisenführ et al., 2010). It is illustrated in Fig.5.3.: We imagine an uncertain alternative A with a 50/50 chance that the value will be 0 or 1. The expected utility EU of such an alternative is $EU(A) = 0.5*0 + 0.5*1 = 0.5$. We now search for an alternative CE (=certainty equivalent) with a certain but unknown value $v(CE)$ so that we are indifferent between A and CE ($EU(CE) = EU(A) = 0.5$). Let us assume this value $v(CE)$ is 0.3. We have now three points of the utility function and can construct more points by applying a similar procedure. The difference between the expected value and the certainty-equivalent is called the risk premium.

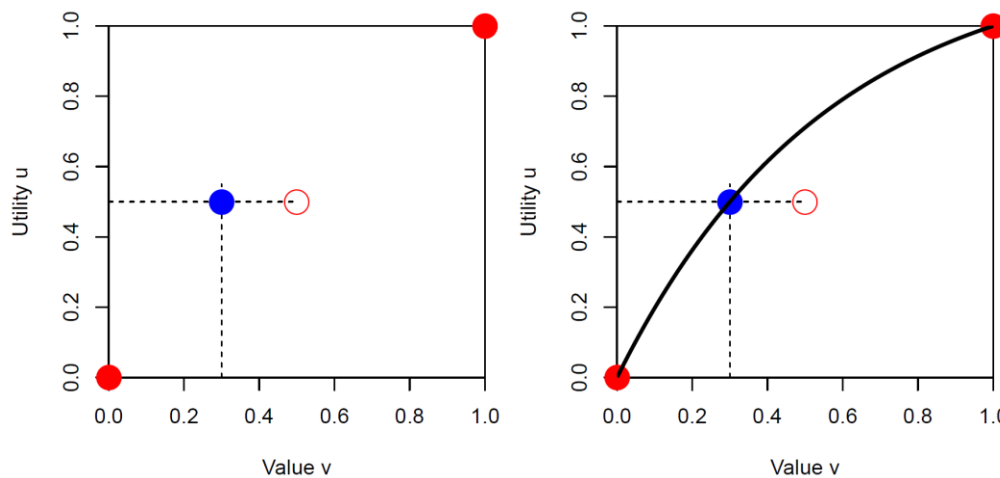


Fig.5.3: Construction of a utility function (solid line). Red dots: Alternative **A** with a 50/50 chance that the $v(\mathbf{A})$ is 0 or 1, $EU(\mathbf{A})=0.5*0+0.5*1=0.5$, blue dot: certainty equivalent with $EU(\mathbf{CE}) = EU(\mathbf{A})=0.5$, $v(\mathbf{CE})=0.3$ in this example (after Schuwirth et al., 2012).

In practice, we seldom have to choose between such extreme alternatives, where some outcomes do not have any uncertainty and others have a chance to perform very bad or very poor. Often, all outcomes are uncertain to some degree. Since the elicitation of risk attitudes is an additional effort, it is advisable to first check, if different risk attitudes (Fig.5.4) would lead to different rankings. If this is not the case, the elicitation of risk attitudes can be omitted and decisions can be taken based on the expected value.

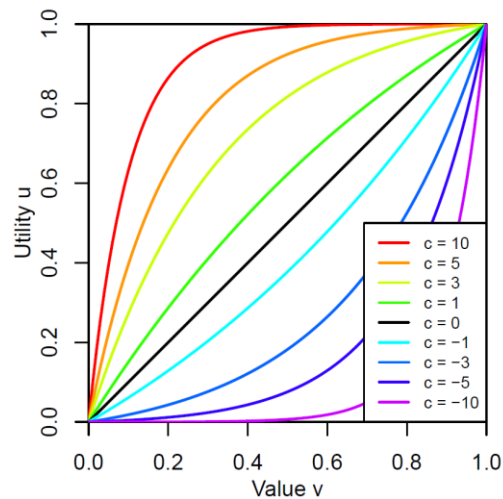


Fig.5.4: Transformation of value to utility functions with an exponential function $u(v) = (1 - \exp(-c \cdot v)) / (1 - \exp(-c))$ and different Arrow/Pratt measures c ; $c=0$ indicates risk neutrality, $c > 0$ indicates risk aversion, $c < 0$ risk proneness.

5.5. Application to the Swiss Case Study

5.5.1 Case Study Site

The study site consists of a channelized reach and a rehabilitated reach at each of the two rivers Thur and Töss. The river Thur at this site is of stream 7th order and has an average discharge of 53 m³/s. The river Töss at the study site is of 6th stream order with an average discharge of 10 m³/s. Figure 5.5 provides an overview of the four sub-sites; more details can be found in Paillex et al. (submitted).

5.5.2 Evaluation of the Rehabilitation Projects

For the implemented projects we have reasonable cost estimates. We assume that the recurring costs (maintenance and routine monitoring) are not significantly different for the channelized reach and the rehabilitated reach. For the comparative evaluation, we can thus focus on non-recurring costs (planning, transaction, land acquisition and construction/investment costs). These are the costs estimates given in Tab.5.1.

The ecological branch of the value function shown in Fig. 5.2 was implemented by first translating the Swiss river assessment protocol (Bundi et al. 2000; <http://www.modul-stufen-konzept.ch>) into a value function, as described by Langhans et al. (2013) for the branch of river morphology. As ground beetles, aquatic vegetation, and riparian vegetation are relevant for the river corridor ecosystem also, the value function of the standard assessment protocol was complemented by branches for these assessment fields (see Paillex et al. submitted for more details).



Figure 5.5: Illustration of channelized sections (left) and rehabilitated sections (right) of the Thur (top) and Töss (bottom) Rivers (pictures P. Reichert, Sept. 24, 2013).

The most difficult part is the quantification of the benefits of rehabilitation for the ecosystem services. As our study on these benefits is still ongoing (Logar et al. in preparation), we have here to rely on the benefits transfer approach (Brouwer 2000; Brouwer and Bateman 2005) to transfer willingness to pay estimates from various sources to the present problem.

The willingness to pay (WTP) for ecological improvements was estimated from the recently revised Water Protection Act, which allocates 5 billion CHF to rehabilitate 4'000 km of the most degraded 15'800 km of Swiss rivers within 80 years. Although this was a decision by the Swiss parliament, we argue that it can be used as a lower bound for the WTP of the population. The reason is that it resulted from a democratic process initiated by the Swiss fishing association, taken over by the parliament, and not questioned by the population (no referendum; see box for more details). This indicates that there is strong public support for this funding. However, as it was not inquired by a survey, we only know that this funding is supported and do not know, whether the WTP of the population would be higher. We thus consider it as a robust lower bound to the WTP for ecological river improvements for river sections not influenced by hydro-power plants (additional funding was decided to reduce adverse effects of hydro-power plants).

Despite having an estimate of the overall WTP for river rehabilitation it is still difficult to distribute this potential funding to rivers of different size. Obviously, the cost of rehabilitating large rivers are higher than those for small rivers. We can thus assume also the WTP to be higher for large rivers than for small rivers. In addition, large rivers have an important bridging function from lower to upper parts of catchments and the fraction of degraded rivers increases with stream order (from about 30% for order 1 to about 60% for order 9). This may even more increase the difference in the WTP between large and small rivers. Rather than assuming the WTP to be independent of stream order (this would lead to a WTP of 5 billion CHF/4000 km = 1250 CHF/m we assume that the same amount of money is invested for each stream order (5 billion CHF / 9) and the WTP per river length is proportional to the stream order. This leads to a WTP increasing from 393 CHF/m for order 1 to 3540 CHF/m for order 9 (2360 CHF/m or 470'000 CHF for the 6th order, 200 m long site at the river Töss and 2750 CHF/m or 4'130'000 CHF for the 7th order, 1500 m long site at the river Thur). Finally, we assume that this WTP corresponds to an improvement by two ecological quality classes which corresponds to a value increase by 0.4.

The canton of Thurgau just recently suggested to spend 28 million CHF for flood protection at another 3.7 km long section of the river Thur (close to Weinfelden) to avoid potential damages of up to 360 million CHF. As it is probable that this budget will be accepted in a democratic process, we can again use it as a rough estimate of the WTP of the population for flood protection. This would lead to a WTP of 11 million CHF for our study site of 1.5 km length. As the current implementation strategy of flood protection includes rehabilitation, we subtract the 4'130'000 CHF that are already covered by the rehabilitation funds. Still, the remaining WTP for pure flood protection of 6'870'000 CHF already exceeds the project costs. We assume that this corresponds to an increase in value from 0.5 to 0.9 (the fulfillment of the goal of reaching a high flood protection level will increase from 50% to 90%; note that we will assume that this will be the actual value change for this objective, so that this assumption cancels out). There is no need for improved flood protection at the river Töss.

Finally, we applied the benefits transfer approach (Brouwer 2000; Brouwer and Bateman 2005) to transfer the estimate of willingness to pay for recreational and aesthetic value at a similar site at the Thur River from Spörri et al. (2007) to our site. In that study, it was estimated that 200 persons would spend 11 CHF on 25 days per year. If we sum this up over a period of 50 years with a discount rate of 3% we end up with 1'430'000 CHF. As the river Töss is considerably smaller and the rehabilitated section much shorter, we assume that there are 10 times less visitors. This leads to a WTP for recreational and aesthetic value of 143'000 CHF. Similar to the case of ecological improvements, we assume that this WTP corresponds to an improvement of the morphological state by 0.4 value units (2 state classes).

Table 5.1 summarizes these estimates. Note that the sum of the estimated WTP in this table corresponds to the value changes also listed in the table and have to be corrected for the actual change in value. In the MCDA this is done by specifying the weights at the top aggregation level. These weights depend on the range over which the value function is defined. For costs we use the range of our alternatives which is equal to the rehabilitation costs (because there are no non-recurring costs when keeping the channelized river as it is). For the ecosystem services and the ecological state, we use the ranges from a very bad to a

very good level of services or state, respectively. For these objectives, we assume that the estimates of the WTP correspond to a value increase by 0.4 (as indicated in Table 5.1). The aggregation weights are then selected to be proportional to the costs or the WTP divided by the value range. This makes the MCDA equivalent to a CBA. Note that for a complete analysis, we would also have to estimate weights for the objectives, the degree of fulfillment does not change by rehabilitating the river (self-purification, groundwater supply, and – for the river Töss – flood protection). As these weights do not influence the difference in value at the highest level, we do not provide estimates and set these weights to zero. This would have to be corrected in a more detailed analysis, but it does not affect the ranking of the alternatives.

Objective	Thur River (1500 m)			Töss River (200 m)		
	Value Range	Costs (M CHF)	Est. WTP (M CHF)	Value Range	Costs (M CHF)	Est. WTP (M CHF)
Non-recurring costs	1.0	6.00		1.0	0.50	
Ecological state	0.4		4.13	0.4		0.47
Flood protection	0.4		6.87	-		-
Recreat. and aesthetic val.	0.4		1.43	0.4		0.14
Total		6.00	12.43		0.50	0.61

Table 5.1: Estimates of non-recurring costs and willingness to pay for different objectives in the objectives hierarchy shown in Fig. 5.2. We assume that the WTP estimates correspond to a value increase indicated in the left column of each case.

Implementation of the value function was done in R (<http://r-project.org>) based on the R packages “ecoval” and “utility” (Reichert et al. 2013), both available at <http://CRAN.r-project.org>). These packages were also used to produce Figs. 5.1, 5.6 and 5.7.

5.5.3 Results

Figures 5.6 and 5.7 show the results of the value functions for the rehabilitation projects for the Swiss case studies at the rivers Thur and Töss with and without uncertainty ranges of the calculated values.

Within the branch of the characterization of the ecological state, Figs. 5.6 and 5.7 show that rehabilitation increases the physical state to the highest quality class. Due to the short rehabilitated section, the chemical state was not expected to change and was measured in the degraded section only. The biological state increases significantly, but as it crosses only one class boundary, it increases only by one quality classes for each of the rivers. Despite the significant increase in many aspects, the ecological state does not quite reach a good state (green color) for the river Thur, but it does for the river Töss. See Paillex et al. (in preparation) for a more detailed discussion.

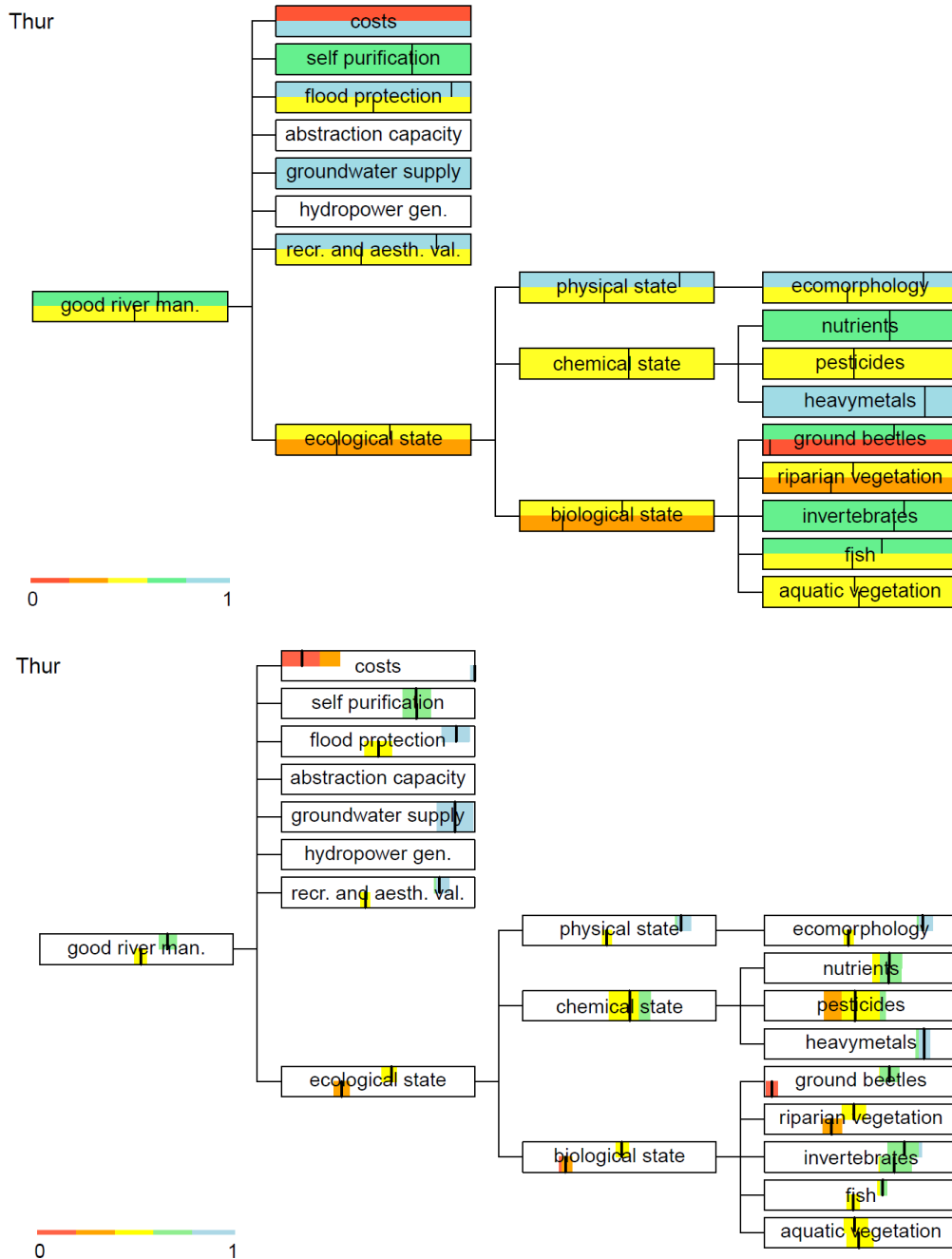


Figure 5.6: Comparison of the evaluation of rehabilitated (top part of the boxes) and channelized (bottom part) sub-sites of the river Thur. Vertical lines indicate values on a horizontal scale from 0 to 1 in each box. Top panel: the sub-boxes are colored according to the five colors indicated in the legend. For the ecological sub-objectives, this corresponds to ecological quality classes. Bottom panel: colored ranges represent 90% uncertainty ranges, vertical lines medians. White boxes indicate that no data is available as these sub-objectives are not applicable at the investigated site.

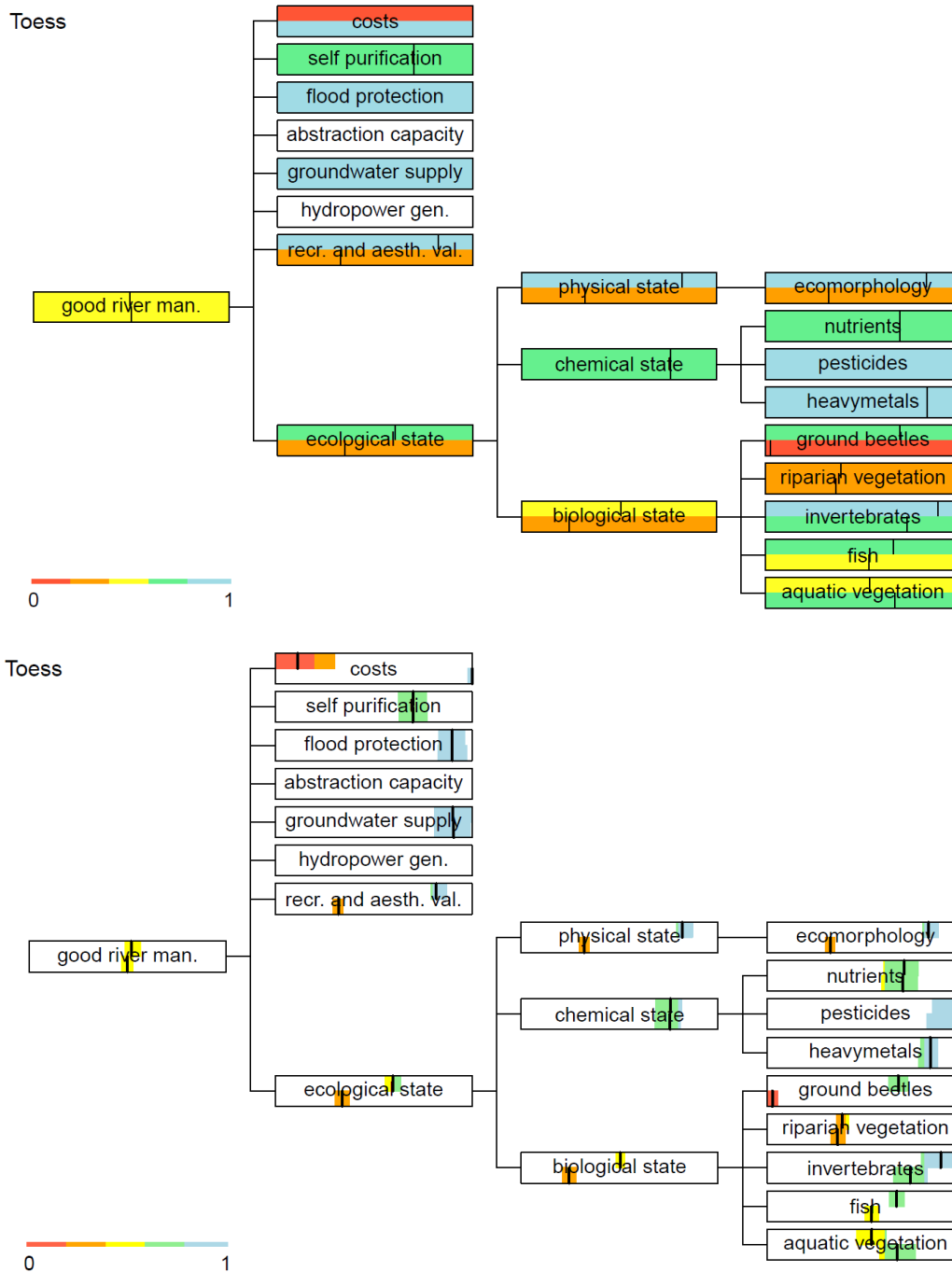


Figure 5.7: Comparison of the evaluation of the rehabilitated (top parts of the boxes) and the channelized (bottom parts) sub-sites of the river Töss. See caption of Fig. 5.6 for explanations.

At the highest hierarchical level, the rehabilitated section has a significantly higher value than the degraded section in the case of the river Töss. For the river Thur, both overall evaluations lead to very similar values, but in the rehabilitated case reaching a good

ecological state. The higher increase for the river Thur is mainly caused by the combined effect of improved ecological state and improved flood protection level and the high willingness to pay for flood protection.

5.5.4 Discussion

This analysis provided important insights into the costs and benefits of river rehabilitation projects and on how a transparent outline of these issues can support transparent societal decision making about the trade-off between costs and benefits.

- The rehabilitation project at the river Töss fulfilled the goal of reaching a good ecological state, whereas that in the river Thur just failed to reach this goal. This is mainly caused by a low chemical state and poor vegetation communities with the presence of invasive plants. These results are preliminary as some of the assessment modules are still at a preliminary stage, and the states of the rehabilitated sections are close to the boundary between the classes of a moderate and a good state.
- The comparison of the overall evaluation clearly demonstrates that combined flood protection and rehabilitation projects make it much easier to outweigh the costs by the benefits.

Our analysis still has some deficits. The three most important ones are the following:

- Willingness to pay estimates for ecological gain and ecosystem services are based on transfer from past studies and the democratic legislation process in Switzerland. This provides a reasonable rough estimate, but more precise estimates could be obtained from a study that would collect primary data. Such a study is currently under way (Logar et al., in preparation). Note that a similar procedure as used here could be used to optimize the gain in ecological state and ecosystem services given a fixed budget or to determine the most cost-effective alternative to reach a certain ecological state. This task would not be affected by the high uncertainties associated with the benefits estimates of ecosystem services.
- Analyzing cost-benefit trade-offs for individual rehabilitation projects bears the deficit that synergies of optimal combinations of such projects by systematic, spatial rehabilitation planning are neglected (Langhans et al. 2014b; Reichert et al. 2015a). As rehabilitation of a whole river corridor at a larger spatial scale can lead to significant additional improvements in particular in the ecological state and for recreation compared to restoring the same total river length dispersed over the catchment, the benefit estimates can be assumed to be closer to a lower bound than to a median estimate. We will try to address these issues in Reichert et al. (in preparation).
- Uncertainty about outcomes of alternatives, about the valuation of ecological outcomes by scientists and of societal objectives by stakeholders or a sample of the population, and the effect of risk attitudes (mostly risk aversion) did not change these results significantly.

6. Conclusions and Policy Recommendations

There are two important **sources of uncertainty** to consider in environmental management in general, and in particular for river restoration:

- **Uncertainty about scientific predictions of outcomes.**
Depending on alternatives, this requires prediction and uncertainty estimation of the behavior of a natural system, natural-technical system, or even of a combined natural-technical-socio-economic system (e.g. in case of measures that include incentives to some of the affected stakeholders). In particular, one has to consider the potential for adverse outcomes as discussed in chapter 3.
- **Uncertainty about the preferences of the society elicited from inquiries or stakeholders.**
In addition to the difficulties of the stakeholders to be aware of their own preferences and to be able to quantify them, this also includes their risk attitude (how uncertainty about the outcomes affects their preferences).

Policy recommendations:

- **Communication of uncertainty is a key element of any communication of scientific predictions.** Visualization of uncertainty ranges can support this task. Lack of communication of scientific uncertainty in the past led to a reduction of trust of stakeholders to scientists.
- **Clearly separating scientific predictions and societal valuations is an essential element of any decision support procedure.** Uncertainties in both elements should be clearly communicated separately. In particular if there are disagreements among experts about scientific predictions and of stakeholder groups about preferences.
- **Uncertainty about scientific predictions can be addressed by probability distributions and scenarios; uncertainty about societal preferences are often better addressed by sensitivity analyses** of the ranking of the alternatives resulting from combining predictions of the outcomes of decision alternatives with preferences.

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