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Evaluation of brine discharge to rivers and streams: Methodology of rapid impact assessment

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ABSTRACT

Induced by a brine discharge study for a submerse gas storage cavern project, a suitable methodology for rapid impact assessment had to be found. In this paper a simple stochastic, stationary model is described for assessment of intensity and temporal variability of chloride pollution at the regional scale of the rivershed. Chloride concentration is used as a proxy of salinity. It is assumed to be the result of deterministic process (flow-dependent) and stochastic variation (estimated for boundary conditions and tributaries by an additive error term based on PERT distribution). This approach is suited to conduct Monte Carlo simulations in order to calculate long-time means and percentiles of the prospective in-stream chloride concentration (exposure model). The biocoenoses exposed to this pollution has to be evaluated in terms of chloride tolerance. Herefore Maximum Field Distributions (MFD) of relevant species (aquatic macrophytes, macroinvertebrates, fish) were compiled and merged to Species Sensitivity Distributions (SSDs). Critical aspects of MFD data quality are discussed. Chloride model simulations representing different discharge scenarios provide exposure parameters (e.g. 90th percentile) that can be compared with SSD-derived protection levels (e.g. maximum loss of 10% of taxa) to quantify and evaluate possible adverse effects as well as potential recolonisation in case of load removal. Crosslinks to conservation issues are relevant in the selection and position of rare or protected species in the SSD. As an analysis of the German legal framework and technical guidelines revealed lack of guidance and best practices for such assessment and impact evaluation, recent experience highlights serious needs in applied research.

Introduction

In the Mid German region, local salt production using natural saline springs started during the early iron age (700–400 y AC, Neuß and Zühlke 1982). Large scale salt-works, potassium and soda industries developed since the middle of the 19th century. Discharge of brine and other chlorine sewage caused significant loss of biodiversity in the receiving rivers. Other water use (e.g. drinking water supply, irrigation, raw water for chemical industries) was impacted adversely (Miersch 1966). During the 1990s, closing of potassium mines and chemical industries reduced chloride pollution substantially. Remaining industries reduced their discharges and produced a less temporally variable discharge. However, open mineral deposits with intensive saline leaching are important emitters of chloride as before. Parallel to this, domestic sewage systems and wastewater treatment were renewed and improved. The recolonisation of heavily damaged rivers by their natural biota started immediately (LAU LSA 1997; Tappenbeck 1998; Gaumert 1998; Schöll and Fuksa 2000) but is not finished yet.

At present the reactivation and expansion of potassium mines, growing soda production and active leaching of subterranean gas storage caverns in Permian (Zechstein) salt deposits will lead to new brine discharges. Due to the recent European Commission (EC) directives and their implementation in national German law (Habitats Directive 92/43/EEC, Water Framework Directive 2000/60/EC, Freshwater Fish Directive 2006/44/EC), environmental and conservational issues got more weight in decision making on such intentions. The German Working Group on water issues of the Federal States and the Federal Government (LAWA) defined a Germany-wide river typology and provided type-specific limits of key parameters for maintaining very good (= reference or near reference) and good (= moderately impacted) ecological condition/potential according to the WFD (LAWA 2007). Such limits are given for annual (arithmetic) means of chloride (50/200 mg L^{-1}) and sulphate (50/200 mg L^{-1}). But these limits shall explicitly not be used for river systems with natural saline impacts. Further being annual mean levels they can give misleading indication of strongly oscillating ion concentrations. At last these limits will not automatically exclude adverse impacts on habitat types and species listed in Annex I, II or IV of the EC Habitats Directive. Wastewaters

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from potassic and soda industries are excluded explicitly from the emission based regulation of the German Waste Water Ordinance. Sound methodological standards, mandatory guidelines and best practices for prediction and evaluation of adverse impacts due to brine discharge are not approved in Germany until now. Applicants for discharge permission and their consultants as well as the competent authorities are compelled to deduce their methodology of impact assessment and ecological risk evaluation every case anew.

In this paper a methodological approach for rapid impact assessment is described, developed within the scope of a feasibility study for a subterranean gas storage cavern project in Saxony-Anhalt, Germany. Further there will be derived some conclusions on data representativeness and research, needed to enable practicians to advanced forecast and adequate evaluation of salinity impacts.

Quantification of emission and in-stream concentration

Expected salinity and other discharges are determined by capacity and technology of the intended project, i.e. they are primarily an economical and technical issue and will not be considered here. Three key issues of exposure assessment have to be clarified first: intensity, temporal pattern and spatial extent of prospective saline pollution in the receiving river sections downstream of the discharge point. The project engineers should be able to supply information on mean and range of the discharged loads as well as temporal variability in discharges which are essential for estimating the result salinity and other concentrations in the receiving water from which impacts on the aquatic environment can be considered by limnologists. If fast and complete mixing is not expected, quantification of spatial extent requires two- or three-dimensional description of the saline effluent plume.

Stochastic chloride model

As salt loads of industrial emitters can vary independently from natural hydrologic variation in the receiving river system, typical anthropogenic saline pollution cannot be described adequately by a single mean or by an arbitrarily selected low flow situation. Thus, calculation of salt concentrations in the receiving water body shall include statistical parameters like arithmetic mean, range, 90th percentile, and maximum, for a time series (e.g. 1 or 10 y). Up- and downstream inputs by tributaries, other pollutants and diffuse natural sources have to be considered. The longitudinal extent of salinisation is determined by downstream dilution to an uncritical level. To meet these requirements, despite restricted time-lines, a simple stochastic chloride model was built using steady-state mixing calculations for the river system under investigation. The simulated system has been designed for

- rivers (Elbe, Saale, Bode), segregated into 37 sections by 34 mixing nodes at all,
- upstream inflow node, one final downstream node (all these nodes represent gauging and quality monitoring stations),
- 8 gauged tributaries (with quality monitoring stations close to the river mouth),
- 6 brine discharge points (real range and means of discharge Q and chloride loads are known for four of them, permitted flow-dependent discharge rules were used for the fourth, and submitted planning values were used for the fifth as well as for the project under study).

All nodes are assumed to be continuously stirred tank reactors (Chapra 1997), and the entire river is modelled as a series of these reactors. A conceptual schematic of the river system is given in Fig. 1.

At every node downstream of a river section $[n]$ the several calculations were done Eqs. (1)–(4). First the concentration of the pollutant, e.g. Cl, at point n is estimated $c_{[n]}$ as:

$$c_{[n]} = \frac{(c_{[n-1]} \cdot Q_{[n-1]}) + \sum(c_{trib[n]} \cdot Q_{trib[n]}) + \sum(c_{disc[n]} \cdot Q_{disc[n]})}{Q_{[n-1]} + \sum Q_{trib[n]} + \sum Q_{disc[n]}} \tag{1}$$

Eq. (1) is simple mixing of all inputs in section $[n]$. Then with Eqs. (2) and (3) the flow balance of the section has to be modified in order to account for diffuse fluxes in broadest sense (in- or exfiltration and small tributaries without gauging). Such fluxes are assumed to be relevant if the sum of all known tributary and point discharge flows does not equal the flow difference between the upstream and downstream gauging station. Every river section gets a share of the flow difference proportionally to the relation of the section's length to the distance between the two gauging sites.

$$Q_{diff_{[n]}} = \frac{Q_{dsg} - Q_{usg} - Q_{trib[n]} - Q_{disc[n]}}{km_{usg} - km_{dsg}} \cdot l_{[n]} \tag{2}$$

$$Q^*_{[n]} = Q_{[n]} + Q_{diff[n]} \tag{3}$$

Eq. (4) will adjust the pollutant concentration if there is some diffuse inflow as indicated by Eq. (2)/(3). If diffuse flow is zero or negative (e.g. infiltration), the pollutant concentration remains the same. In case of positive diffuse flow, an additional mixing step will be done.

$$\begin{aligned} &\text{If} \quad Q_{diff[n]} \le 0; \quad c^*_{[n]} = c_{[n]} \\ &\text{If} \quad Q_{diff[n]} > 0; \quad c^*_{[n]} = \frac{(c_{[n]} \cdot Q_{[n]}) + (c_{diff_{[n]}} \cdot Q_{diff_{[n]}})}{Q^*_{[n]}} \end{aligned} \tag{4}$$

c = concentration of pollutant [mg L^{-1}]	Indices:
c^* = concentration of pollutant, adjusted for diffuse flux [mg L^{-1}]	$[n]$ = river section under calculation
Q = flow rate [m^3 s^{-1}]	$[n-1]$ = upstream river section
Q^* = flow rate, adjusted for diffuse flux [m^3 s^{-1}]	*disc* = pollutant discharge
km_{usg} = river stationing at upstream gauging station [km]	*trib* = natural tributary
km_{dsg} = river stationing at downstream gauging station [km]	*diff* = diffuse flux
$l_{[n]}$ = length of river section under calculation [km]	

These calculations can be done in parallel for several conservative ions of dissolved mineral salts (e.g. SO_4^{2-}, Cl^-, Mg^{2+}, K^+, Na^+, ...). Here only chloride is used as guiding parameter for NaCl-dominated brines. Traditionally most permits for brine discharge in Germany refer to chloride and sometimes additionally to other parameters, e.g. total hardiness.

For all upstream inflow nodes and monitored discharge points, Q/[Cl^-] – relations were calculated as power functions [Cl^-] = $a + Q^b$, where a and b are parameters to be estimated. For some minor tributaries and discharge points with nearly constant chloride concentration, simply [Cl^-] = constant was set. All pairs (Q, [Cl^-]) were recalculated. Then, after checking for trend, PERT distributions (Clark 1962) were fitted to the residuals. PERT is an acronym for "Program Evaluation and Review Technique", indicating the origin in project management. This distribution was "rediscovered" in the recent past for use in hydrology and in ecological modelling (e.g. Lowell and Benke 2006; Overholtz and Link 2007). The PERT distribution is a special case of the beta-distribution, defined only by minimum, most probable value and maximum (Vose 2002). It uses them to create a smoothed curve that fits well to any normal or lognormal distribution. Here it was chosen due to its flexibility and simplicity in use.

The model is driven by a 10 y-series of daily discharge for all rivers and tributaries. Daily chloride concentration for all river's inflow nodes, all monitored tributaries and all discharge points is

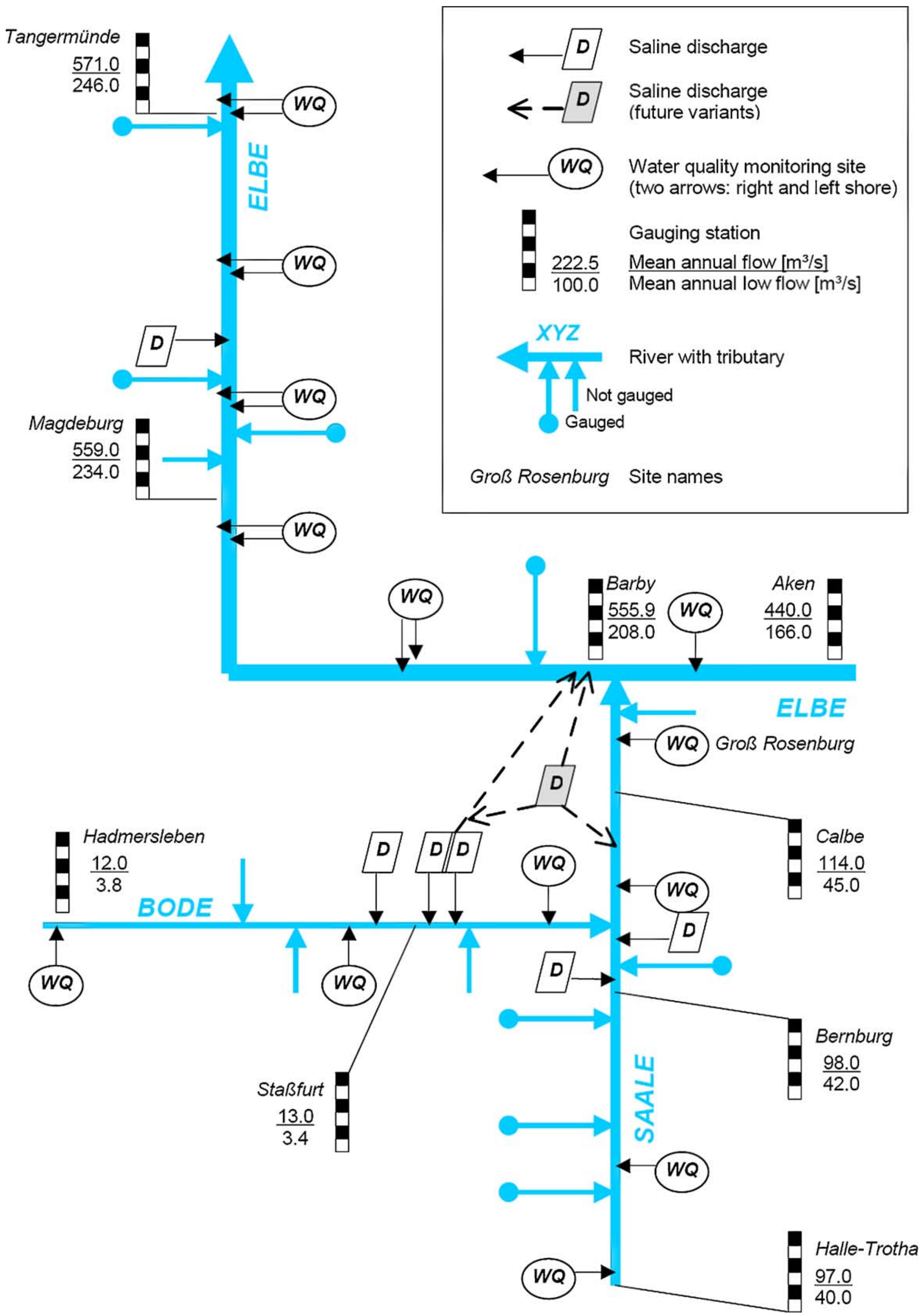

Fig. 1. Chloride model: conceptual schematic of the river system under study.

calculated by $[\mathrm{Cl}^-] = a + Q^b + E$ or by $[\mathrm{Cl}^-] = \text{constant} + E$. The error term E is a random variable, following the PERT distribution of the residuals.

For any given date, the measured daily flows are supplemented by chloride concentrations calculated as aforementioned. Then in all model nodes mixing calculations are done under the assumption of stationarity. A Monte Carlo analysis was set up in the way that the dates are repeatedly selected by random as well as the error terms E. Doing so, 3652 replications simulate a 10 y series. Means, percentiles, etc., are extracted from the resulting chloride

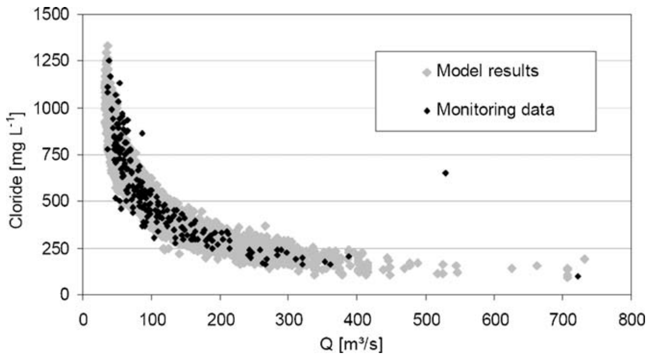

Fig. 2. Validation of a simulated 10 y series of chloride concentration (Saale River, Groß Rosenburg) Monitoring datasets 1995–2005 from Landesbetrieb für Hochwasserschutz und Wasserwirtschaft Saxony-Anhalt, 2008.

data series. Due to the simple structure of this model it was possible to implement it using MS EXCEL.

Model validation

A validation was done by superposing calculated data with measured data from monitoring stations which were not used in model calibration. For this purpose the near-mouth monitoring stations of the Saale and Bode rivers and the last monitoring station at the lower boundary of the Elbe section under study were retained. An example is given in Fig. 2. The model does not account for dynamic aspects (pollution wave, longitudinal dispersion) but gives very realistic estimates for mean and range of chloride concentrations over a long period (Table 1). Scenarios with variation of loads, discharge sites and transfer of brines to other river stretches facilitate assessing both load removal and rising load in distinct sections of the whole river system.

Estimation of plume characteristics

The stochastic chloride model was set up under assumption of complete mixing. Before an effluent plume is completely mixed laterally, it will form zones with much higher concentration within the cross section. The remainder of this effect can bee seen in Table 1: right/left shore measurement at Tangermünde monitoring site reflect still the decelerated lateral mixing of the chloride polluted Saale river (LAU 1998), a tributary about 95 km upstream. Thus, the spatial extent of effluent plumes may be critical for distinct structures and functions in the receiving fluvial ecosystem. For instance, it could cover spawning areas or essential habitats for juveniles of native fish species within the main channel. The plume could form a chemical barrier in front of tributary mouth, which may be a preferred migration path fish or provide refuge habitats for other mobile species. Finally water extractions for other uses (process water, irrigation) could be affected by saline water.

A state-of-the-art numerical 3D-simulation (Ji 2008) would provide detailed data on extent or shape of and concentration gradient within the effluent plume. However, for screening level assessments and feasibility studies, the time and resources usually does not justify such simulations. It is better to put effort into the analyses of conceptual alternatives of the project, reliable scenarios, and proper definition of boundary conditions. Data extensive modelling should be replaced by simplifying approaches from the hydraulic engineer's toolbox. Assuming an uniform rectangular cross section for the navigable channel between the groynes of the Elbe river under low flow conditions, it is an exercise of ease to apply a coarse 2D advection-dispersion-model for conservative substance (Fischer et al. 1979; Baumgartner et al. 1993).

As the distance of the discharge point from the nearest shoreline can be varied, such a simple model can already be used to evaluate technical options. Discharge of pollutants via a side channel will form a very long and narrow lateral plume whereas an in-stream diffuser, e.g. mounted at a bridge pilar, will force a faster lateral mixing. An example is given in Fig. 3. Further superposition with more chloride sources is possible and necessary in the context of cumulative impact assessment as demanded by the EC Habitats Directive. In terms of variant evaluation and mitigation, one will check the downstream river section for important structures, tributaries and habitats to decide about the less adverse impact: maintain a less polluted part of the channel accepting a prolonged zone of higher concentration close to the opposite shoreline (Fig. 3A) or faster achievement of final mixing concentration on both sides of the river (Fig. 3B). Here the further assessment must apply regulatory mixing zone approaches although there is no explicit legal guidance for this subject in Germany (Bleninger et al. 2004).

Tolerance of aquatic biota

Taxa selection and established bioindicator systems

Despite the rich literature on salt pollution and natural estuaries there has not been set up an official comprehensive database of salinity tolerance of freshwater biota in Germany. According to the biological quality elements requested by the WFD and the habitat types/species protected by the EC Habitats Directive, lampreys and fish (further together in short: fish), macrozoobenthos, and aquatic macrophytes are relevant groups for impact assessment and their salt tolerance was investigated.

The few salt-related bioindicator systems used Germany comprise the diatom based Halobion System (Ziemann 1971; Ziemann et al. 1999), and a macrozoobenthos-based salinity tolerance classification within the PERLODES system (Meier et al. 2006). Both systems are designed for a posterior evaluation of salt pollution. Similarly, Bäthe (1995) sorted the macroinvertebrates of the salt polluted Weser into tolerance classes with boundaries derived from Remane's (1971) classification of brackish waters. Preference levels or tolerance classes used by the above mentioned authors have a relatively coarse resolution with no subdivision of the (β-)oligohaline zone (S = 0.5‰ . . . 3.0‰; Remane 1971; or S = 0.5‰. 5.0‰ of the VENICE System; Anon. 1958). However, in middle and central European inland waters, one may expect significant differences in aquatic communities between the upper and lower boundary of this zone. Contrary to recent developments, e.g. in Australia (Horrigan et al. 2005, 2007; Dunlop and McGregor 2007) the indicator systems mentioned above may give some orientation but are not sufficient for predictive impact assessments. Every practitioner has to compile and evaluate available species tolerance data. This is somewhat subjective and results in different intensity and different outcomes of the efforts. The results presented here are a product of such attempt, too.

Pragmatic approach: Species Sensitivity Distributions (SSDs) based on empirical field data

For most taxa recently found in the potentially impacted river sections Maximum Field Distribution (MFD, Kefford et al. 2004) was compiled using white and grey literature research and monitoring data at hand. A MFD describes the maximum stressor intensity under that a certain species was recorded in the field. The MFD of a species along a salinity gradient, measured by chloride concentration, is used as a proxy for the true salinity tolerance of the species. Salinity tolerance of a biotic community can best be visualised as fraction of the local species pool tolerating a certain

Table 1
Simulated 10y series of chloride concentration (mg/l): Validation od model results. Monitoring datasets 1995–2005 from LHW LSA, 2008; n = number of samples.

River	Bode		Saale		Elbe		
Site	Neugattersleben		Groß Rosenburg		Tangermünde		
Year	2000–2004		1995–2005		1995–2005		
Cl [mg/l]	Measured	Modelled	Measured	Modelled	Measured, near left shore	Measured, near right shore	Modelled
n	77	1644	210	3652	206	201	3652
Max	3810	6738	1250	1328	320	280	297
90th percentile	2812	2986	873	865	220	198	215
Mean	1520	1588	542	530	146	137	150
Median	1310	1333	499	485	132	126	146
10th percentile	456	506	263	248	78	84	86
Min	171	137	97	93	44	42	36

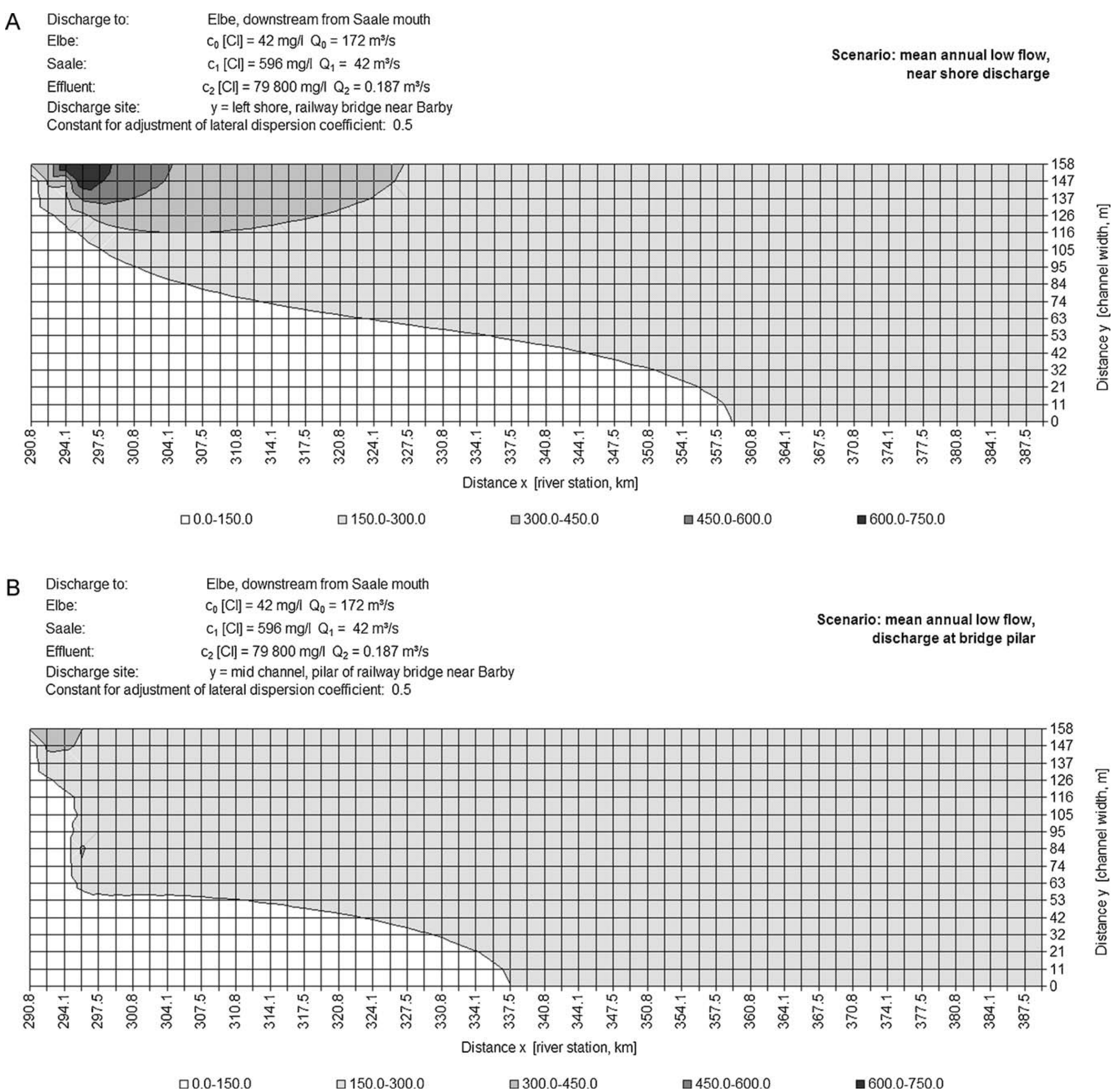

Fig. 3. Formation of the effluent plume with discharge site downstream from a preloaded tributary (Cl^{+} mg L^{-1}). (A) Near-shore discharge site and (B) mid-channel discharge site Elbe River below mouth of the Saale River. Approximate mixing calculation according to Fischer et al. (1979). Notice the differing scales of x and y.

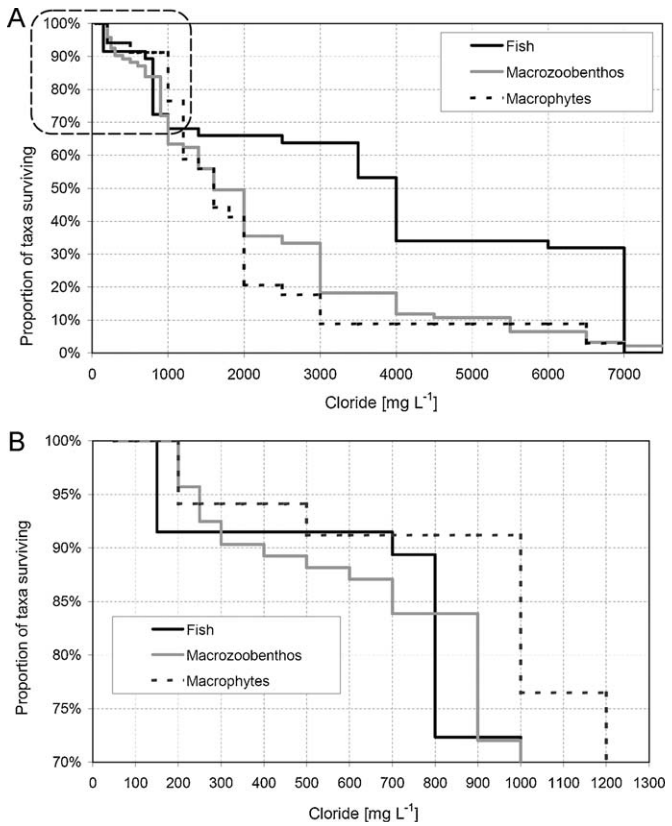

Fig. 4. (A) Chloride tolerance of riverine biocoenosis: SSD based on pooled MFD data. Kaplan–Meier function. (B) Detail of (A), range below 1000 mg L^{-1} Cl^-.

concentration of chloride as indicated by their MFDs. Such fractions plotted against increasing concentration levels give the Species Sensitivity Distribution (Kooijman 1987). Empirical SSDs for the fish, macrozoobenthos and aquatic macrophytes in the river system under study are presented in Fig. 4 using the Kaplan–Meier function (Kaplan and Meier 1958). Comparison of the local biocoenosis' SSD and modelled salinisation parameters describing intensity and likelihood of exposition makes it possible to forecast adverse impacts due to future saline pollution (Hart et al. 2003).

Kefford et al. (2005) objected that SSDs are frequently constructed using to few species. So it is uncertain whether given percentage in an SSD will protect the same percentage of species in nature. As different taxonomic groups are recorded with varying intensity and efficiency, this objection can be cleared here for fish, were all 47 taxa of the potential native reference fish communities (IfB 2006) could be attributed with a MFD value. Further work is needed in macrozoobenthos and macrophytes, e.g. inclusion of rare species that are underrepresented in the standard monitoring data or that are absent today but potentially hampered in their future recolonisation. Recently, Eggers (2006) reported 90 macrozoobenthic taxa from two sampling areas at the Middle Elbe whereas the extensive compilation in Schöll and Fuksa (2000) includes at least about 250 species (but with most Chironomidae determined at species level) for the metapotamon of the Elbe. In contrast to these figures, the average routine benthos sample from the middle Elbe contains about 16.8 taxa (min = 7, max = 35, n = 13, unpublished data GLD Sachsen-Anhalt, 2008). The SSD for macrozoobenthos in Fig. 4 is constructed from 101 taxa for which plausible MFD data could be found. For aquatic macrophytes, the SSD was derived from 34 of 48 species which form the typical aquatic associations of running waters and backwaters of the middle Elbe region (Hilbig et al. 1987; Täuscher 2000).

However, the frequently criticised (Newman et al. 2000; Forbes and Calow 2002; Kefford et al. 2005) construction of SSDs using a handful of species is less relevant here, as the SSD used represents still a large fraction of the real communities. The assumption that the species with tolerance data are representative also for species without available data remains to be verified in future work. Given the large proportions of the communities represented by the use of MFD, it is relative contribution to bias in risk assessment is surely very much reduced.

Use and interpretation of heterogenous data

To avoid the potential errors caused by spot chloride measurements when salinity is temporally variable, where possible the 90th percentile of the annual chloride concentration at the site and year where the biota was recorded was used. This is consistent with the German pollutant specific water quality class boundaries according to LAWA (1998) and has inherently a safety-application factor if used as MFD boundary. Unfortunately, many published records of species occurrence do not allow this parameter to be determined. Then, two options are possible: to retain possibly misleading information (but have MFD values for more, especially rare species) or to discard data (but have MFD values for few species). Mostly the first option was selected as lack of a statement on rare or keystone species in a permit application is much more critical in terms of EC and German conservation law than a raw conservative estimate. The records in question were used following these rules:

(1) Less than 11 chloride measurements for a species are available so that the 90th percentile cannot be calculated: the second value after sorting in ascending order was used like the 90th percentile.
(2) Only annual chloride mean is reported: also used like the 90th percentile. Very conservative assumption (accepting type 1 error in risk evaluation).
(3) Anecdotal species record with only one chloride measurement: accepted and used like the 90th percentile. Probability of being above the 90th percentile based on monthly (n = 12) sampling is p = 0.167, probability of being below is p = 0.833, i.e. potential error tends much more to a conservative, protective result (i.e. accepting type 1 error in risk evaluation) than to overestimating the true tolerance (type 2 error in risk evaluation).
(4) Annual mean and maximum salinity of the sampling site is reported: the arithmetic mean of both parameters was used like the 90th percentile. From 38 annual time series of more or less salt polluted German river sections, the result of this operation underestimated the 90th percentile in 84.8% of all cases. So it is plausible to assume that potential error tends more to a conservative, protective result here, too.
(5) Species attributed with a salinity class according to the VENICE system or a similar classification: the arithmetic mean between the class boundaries was used, e.g. β-oligohaline $0.5‰ \leq S \leq 3.0‰ \rightarrow S = 1.75‰ \approx 953\ \mathrm{mg\ L^{-1}}$ Cl^- assuming $S \approx 0.0018 \times Cl^-\ [\mathrm{g\ kg^{-1}}] - 0.028$ (Schlieper 1971). The same was done in case of species records from estuaries, that could only be attributed with the local annual amplitude of salinity (e.g. Stettiner Haff: $0.8‰ \leq S \leq 2.4‰ \rightarrow S = 1.6‰ \approx 870\ \mathrm{mg\ L^{-1}}$ Cl^-). Overestimating the true tolerance (type 2 error in risk evaluation) is possible because it cannot be excluded that the species use only a salinity range close to the lower boundary.

To sum up may be said, the data used to calculate MFD values are heterogenous in terms of taxonomical resolution, geographic origin (but restricted to Europe), type of water body and related salinity data aggregation. This heterogeneity may lead to bias of results in both over- and underestimate of salinity tolerance. Since this fact is crucial for reliability and acceptance of further conclusions some of this issue will be adressed in detail. The origin of tolerance data can be classified as follows:

- primary monitoring data bases provided by environmental agencies, mostly with taxa lists from biological sampling and corresponding physicochemical datasets, covering a wide salinity gradient by geogenic salt load or anthropogenic pollution,
- description of freshwater species records within estuaries, the latter characterised usually by isohalines and/or annual range of salinity for the sampled sites,
- as a last resort, previous compilations based of one the above mentioned data types or published lab-test data were used if no other data for a species were available. In such case, a critical evaluation of plausibility, reliability and geographical reference of the data is eminently important.

Compilation of primary monitoring data bases is a widely used approach (Kefford et al. 2004; Horrigan et al. 2005; Rutherford and Kefford 2005). Data of this type available in Mid Germany cover the recent recolonisation of salt polluted rivers since onset of load reduction due to industrial restructuring, improved wastewater treatment and flow-dependent discharge management. Retrospectively this data may be seen as the outcome of a large disturbance-recolonisation-experiment but with worse test design since they reflect substantial variation in other pollutants and in pollution history.

As the specific sampling success depends on the frequency of the sampled taxa, rare species or species with spatially or temporally limited occurrence will inevitably be missed in many routine monitoring surveys. Similarly, the possible occurrence of salt tolerant species could be limited due to other stressors, e.g. oxygen deficiency, toxic organics, thermal pollution, lack of suitable habitat structures or recolonisation paths. Non-detected species, even though present or potentially vital under the given salinity, are false negatives and contribute to an underestimate of the real tolerance of the community (Rutherford and Kefford 2005). Also statistical artifacts due to sampling effects of rare taxa and corresponding pollutant concentrations, following a log-normal or other right-skewed distribution, will impair MFD compilation and SSD construction. As shown by Stackelberg and Menzie (2002), this effect tends to produce a conservative underestimate of the true tolerance of the species.

As mentioned above, the converse bias is possible, too. Temporary occurrence of non-tolerant species at saline sites can be due to drifting individuals originating from less saline upstream sections (Hübner, 2007), or due to sporadic colonisation attempts at the sampling site without any chance to complete a full reproduction cycle. On the other hand, the indication on its own that an active mobile freshwater species is able to migrate into an estuary makes it plausible to exclude a particular sensitivity to moderate temporal salinity alteration.

An issue of concern is the taxonomical resolution of both tolerance data and data of the communities for which impacts have to be forecasted (Metzeling et al. 2006). The recent German standard sampling protocol for macroinvertebrates (PERLODES, Meier et al. 2006) stipulates determination according to a standardised taxa list, representing the lowest practical level (Haase and Sundermann 2004). To be consistent within the standard sampling protocol, tolerance data and taxa lists of the target biocoenosis have to be adjusted for the same taxonomical resolution standard. But if conservation and biodiversity matters, one should work as far as possible at species level. Maybe this issue has low relevance for functional aspects, because closely related species are likely to represent similar guilds or trophic levels. However, in detail rare and/or protected species at the sensitive tail of the SSD have to be considered thoroughly. Finally, bias in both directions is possible by genuine taxonomical problems, e.g. splitting up of taxa that were assumed to be one species before, or simply by erroneous determination. This issue has a strong temporal component. As a rule of thumb, such taxonomical variations can be found frequently in the years after new keys or monographs were published. Thus, sorting older records by "pre- or post key origin" may be a helpful option in critical data evaluation.

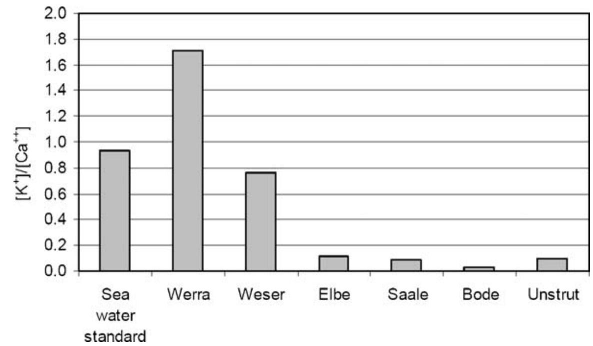

Fig. 5. Mean K^+/Ca^{2+} relation of selected German rivers compared with sea water standard Sea water standard after Schlieper 1971; Werra River near Gestungen 1997, Weser River near Hemeln 1997, datasets by FGG Weser http://www.fgg-weser.de/download_daten.html; Elbe River near Wittenberg 1997, Saale River near Wettin 1997, Bode River near Hohenerxleben 1997, Unstrut River near Freyburg 1997, data from LAU LSA (1998).

Chloride concentration is used in Germany as principal indicator of saline pollution and as an equivalent for comparison with brackish waters in natural estuaries. However, some industrial effluents have aberrant ionic composition compared to natural NaCl-brine or sea water. In detail, enhanced concentrations of Mg^{2+} and/or K^+ were found to be critical for freshwater biota (Albrecht 1954; Schmitz et al. 1967; Ziemann et al. 2001) whereas Ca^{2+} is less critical and can rise up the effect threshold in saline or brackish waters (Schlieper 1971). Variations in accompanying anions like sulphate or carbonate/bicarbonate seem to be less important in NaCl-dominated waters, Zalizniak et al. (2006). Comparing the K^+/Ca^{2+} ratio using anthropogenically polluted German rivers and using sea water as a standard, one can find that only the most intensively polluted Werra River has a conspicuously more K^+ than sea water. Other rivers have a lower K^+/Ca^{2+} ratio than sea water (Fig. 5). So it seems to be acceptable to pool river monitoring data with the additional information gained by inspection of estuary communities instead of discard them due to putative deviant ionic composition. In contrast, MFD data from rivers polluted by the Werra-type of effluents (mainly potassic industries) reflect the "normal" osmotic effect of salinity plus enhanced toxicity due to excess potassium. Thus, these data underestimate the pure salinity tolerance and contribute to an "inherent safety factor" of the resulting SSD if applied to "normal" saline pollution. Conversely, the use of SSD based on MFD data collected in "normal" saline polluted rivers (mainly rock salt and soda industries) and/or natural estuaries will probably underestimate the effect of Werra-type effluents.

Laboratory experiments under controlled conditions would provide most reproducible results for salinity tolerance of single species. Sensitivity of macroinvertebrates derived from acute toxicity experiments reflects sensitivities obtained from MFD data (Horrigan et al. 2007). However, such experiments are in some cases very complicated. Problems arise for instance if the early aquatic stages of merolimnetic insects have to be tested, but exact determination at species level is only possible using imagines (e.g. many Diptera, Plecoptera, Coleoptera, etc.). The effort will increase vastly if species-rich communities are within the scope. Since lab testing is always time-consuming and costly, it is no real option within the scope of feasibility studies and preliminary impact screening in commercial consultancies.

Evaluation of impacts

Maintaining high native biodiversity seems to be the most sensitive feature of streams and rivers exposed to anthropogenic pollution. As proven for moderately and critically polluted rivers, ecological processes like primary and secondary production, microbial decay of organic matter, etc., continued or rise despite the taxa numbers of fish, macrozoobenthos, etc., species considerably reduced. However, loss of biodiversity is in public focus, meets the interest of non-government organisations (NGOs) and fishing associations and can be communicated impressively to decision makers and to the general public.

Fig. 4 indicates the prospective diversity loss with increasing salt pollution. According to a widely used practice in the USA (SETAC 1994), an acceptable level of pollution will be maintained if the 90th percentile of pollutant concentrations does not exceed the critical concentration for the 10th percentile at the sensitive tail of the SSD, i.e. 90% of the species pool will be protected. The 90th percentile of exposure can be obtained from a stochastic model as described above. Diversity losses up to the 10th percentile (protection of ≥90% of species) may be expected for macrophytes at $[Cl^-] \leq 1000\,mg\,L^{-1}$, for macrozoobenthos at $[Cl^-] \leq 400\,mg\,L^{-1}$ and for fish at $[Cl^-] \leq 750\,mg\,L^{-1}$.

As the macrozoobenthos prove to be the most sensitive group, $400\,mg\,L^{-1}$ Cl^- could be set here as preliminary limit for the prospective pollution. In the recently chloride polluted lower Bode river (90th percentile > $2800\,mg\,L^{-1}$ Cl^-, see Table 1), salinity reduction to a level of about $800\,mg\,L^{-1}$ Cl^- would initiate the recolonisation by native fish communities and substantial increase of taxa number. Consequently, a bypass of excess brine of existing soda industries and planned gas storage cavern leaching from the Bode watershed directly to the Elbe River downstream from the Saale River confluence was proposed. In that case, moderate impact on the recent communities in the Elbe River (where all chloride from the whole catchment arrives anyway) would be the cost for substantial improvement in the tributaries.

The selection of a protection threshold – here 90% of species – remains a haphazard valuation without scientific rationale (Forbes and Calow 2002), unless functional aspects of ecosystem process are considered. For the Weser River, Bäthe (1995) demonstrated remarkable shifts within the benthic food-chain due to the loss of many secondary consumers and the beginning reorganisation of the trophic relations when the chlorinity fell below $800\,mg\,L^{-1}$ – indicated by resettlement of native Ephemeroptera, Trichoptera and Bivalvia (Bäthe 2000). Nevertheless, also a functional reasoned "percentile-protection", maybe graded for different ecological guilds, will not match the stricter conservational view on biodiversity loss. The issue is political and has to be discussed amongst the stakeholders. If simple binary statements are inevitable ("the prospective impairment will be significant or not"), it is also inevitable to set up well reasoned regulatory guidelines for such simplification and for the responsible handling of the inherent uncertainty.

The maintenance of a diverse biocoenosis and ecosystem process will not necessarily provide global protection for all objectives of conservation due the EC Habitats Directive. Here several characteristic species of Annex I habitats and species listed in the Annexes II and IV have to be considered. Some of them that are recently not detected by standardised monitoring sampling but proven to be present in some sections of the Elbe River, its backwaters and tributaries. If these species are ranked in the lower tail of the SSD, this position indicates low tolerance – if not biased by false absence (Rutherford and Kefford 2005). At least in such situation it would be counterproductive to ignore well documented lab test results for rare species if the MFD-based SSD is used to evaluate, e.g. an predicted increase in salinity from 200 to $400\,mg\,L^{-1}$ Cl^- whereas the salt tolerance of this species was tested successfully up to $1000\,mg\,L^{-1}$ Cl^-.

A further problem is that by implementation of WFD and Habitats Directive not only impairment of recent overall ecological status, biocoenoses and species has to be quantified. Moreover, the future possibility of achievement of the directive's aims has to be evaluated. Of course an upper boundary of acceptable saline pollution in relation to highly aggregated indices of ecological integrity can be estimated applying "Index sensitivity distributions" in analogy to the above mentioned SSD.

But in terms of the Habitats Directive this means forecasting a more or less favourable conservation status of communities formed by species that are recently rare or locally extinct. Habitat modelling (Guisan and Zimmermann 2000) provides some suitable tools to model potential distribution of such species in future. The SSD can be used then as environmental filter linking predicted exposure data with the model outcome. The focus of such attempt will be on presence-only methods (Elith et al. 2006; Brotons et al. 2004; Tsoar et al. 2007) because these species are predominantly known from anecdotal historical records and/or old scientific collections.

Conclusions

It is possible to conduct a rapid assessment of brine discharge scenarios from the scratch using some well-known tools to estimate intensity, variability and extent of exposure. Comparisons of these estimates with MFD based SSDs of exposed biocoenoses allow for quantification of potential impacts. The approach provides help in decision making in the planning phase of feasibility studies and beyond. Although a mix of few sciences and much pragmatism, the above considerations show that adequate assumptions and deliberate forecasts can be made using freely available and relatively simple tools. Further, the issues discussed tackle more or less directly the existing lack of guidelines and databases needed for predictive impact assessment and evaluation. Salinity is just one of several water quality parameters afflicted with such unsatisfactory arrangement. It may be a bit strange for foreign members of the scientific community that in Germany retrospective evaluation of ecological quality and prospective impact assessment diverge so far from another: sophisticated sets of highly aggregated and intercalibrated metrics as results of long lasting research (e.g. Meier et al. 2006) in contrast to few chemical quality classes, softened limits with exceptions and abstract legal requirements providing much more play for lawyers than for scientific judgement. The practical view on the disproportion in research between retrospective evaluation and causal prognosis helps defining recent needs in applied research:

(1) Completion of empirical SSDs using MFD data for biocoenoses of those German river types which are prospectively exposed to new or enhanced saline pollution; under maximum utilisation of the heterogenous structured and widely dispersed databases, including official water quality monitoring data collected by the federal states and data obtained by WFD research under public funding.
(2) Establishment of sound definition of regulatory mixing zones in Germany (maybe graded according to conservation priorities and to competing water uses downstream).

(3) Development of reasonable guidelines for impact forecast and evaluation, integrating both aquatic ecosystem function and conservation of biodiversity.
(4) More dialogue between the scientific and the permission communities: dealing with uncertainty, separate the possible and well-founded from the questionable (e.g. forecast and evaluation of potential impacts on recently absent species that could or should be present under the far away "good ecological status" or "good state of preservation").

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