A Screening Level Ecological Risk Assessment and ranking method for liquid radioactive and chemical mixtures released by nuclear facilities under normal operating conditions

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INTRODUCTION

Ecological risk assessment process which includes problem formulation, exposure and effects analyses, and risk characterization is commonly used to quantify the potential impact to the environment due to emissions of pollutants from industries. In Europe, the technical guidance document for chemicals (EC, 2003) and recently the ERICA integrated approach for radionuclides (Beresford et al., 2007) recommend to implement, in a first step, a Screening-Tier (or Screening Level Ecological Risk Assessment, SLERA). SLERA is used to evaluate whether the emissions can put the receptor ecosystems at risk or not. Beyond this ERA-type approach, one major challenge still remaining is to gain the capability for assessing radiological impact in a comparative unbiased way to what is done for other stressors such as chemical substances.

Concerning releases from nuclear facilities under authorization, any SLERA is a challenging task because of (1) the large number of substances, (2) the various quantities that may be emitted to the aquatic ecosystems and (3) the various environmental situations to be considered. This task must be performed for two categories of pollutants, radionuclides and chemicals, each exhibiting specificities in terms of concentration in the exposure medium (- or dose) -effect relationships. We describe here the screening and ranking method that has been developed and its first application to the electronuclear sites along the Rhône River (France).

BASIC LINES OF THE CONCEPTUAL APPROACH

The method conceived is largely inspired from methods used to calculate the ecotoxicological impacts in freshwaters within Life Cycle Assessment. It comprises a fate-analysis step described by a fate factor (calculation of the change in exposure from a given release) and a effect-analysis step described by an effect factor (calculation of the change in effect per unit change of exposure) (Pennington et al., 2006). Ecotoxicological exposure-response is mostly based on the Species Sensistivity Distribution theories (SSD) and the potentially affected fraction (PAF) of species as indicator of ecosystem damages (Pennington et al., 2004). The PAF value expresses the toxic pressure put on ecosystems due to the presence of one
chemical, or more realistically a mixture of chemicals, referred as multisubstances PAF or PAFms (Van De Meent and Huijbregts, 2005). The underlying theories and assumptions, plus a number of options (e.g., distribution law) have been extensively discussed elsewhere (e.g., Udo De Haes et al., 2002).

**Fate Factor.** For a screening level assessment, simple transport models in watercourses can be used to describe the increase in environmental concentration per unit of emission for a given substance. A simple box-type dilution model was used to estimate concentrations of a given substance in water. Interactions with sediments were taken into account by using the Kd concept (in L/kg). For a fixed time period (e.g., annual basis), an homogeneous section of a watercourse (mean river flow rate $Q_r$ in $m^3/s$) receiving a quantity $Q_i$ (in mole) of a substance $i$, the average concentration in water ($C_{i,w}$ in mol/L) and in sediment ($C_{i,s}$ in mol/kg) can be expressed as follows:

$$C_{i,w} = \frac{Q_i}{Q_r \times 3.1536 \times 10^6}$$

and

$$C_{i,s} = K_d \cdot C_{i,w}.$$ 

**Effect Factor.** The effect factor addresses the increase in effect per unit of substance concentration. The approach selected here is based on the use of the Hazardous Concentration affecting 50% of species ($HC_{50}$) at their 50% effect ($EC_{50}$) and on the concept of the change in the potentially affected fraction (PAF) of species that experience an increase in exposure above a specified effect level, in the presence of multisubstances (PAFms). For a substance $i$, the linear gradient below $HC_{50i}$ constitutes one of the key assumption adopted giving an effect factor as follows $EF_i = \frac{\Delta PAF_i}{\Delta C_i} = \frac{0.5}{HC_{50i}}$. Estimating the effect factor from the median effect level ($HC_{50}$) is recommended as the best approach when one wants to compare different impact categories (e.g., toxics, eutrophication, acidification) in a final score expressing the sum of all impacts on the considered ecosystem, and enabling the calculation of the relative contribution of each stressor to the total impact (Pennington et al., 2006). The impact due to the multisubstances mixtures is then: $\Delta PAF_{ms} = 0.5 \sum \frac{\Delta C_i}{HC_{50i}}$.

**Working endpoint calculation $HC_{50}$ and $HDR_{50}$.** According to Payet and Jolliet (2004), a minimum data set of three different taxa, preferably chronic $EC_{50}$ on morbidity and reproduction endpoints, is required for the calculation of the $HC_{50}$ in order to cover a wide spectrum of biological responses to the toxic.

For radioactive substances, the primary ecotoxicity values are the Dose Rates associated with a 50% Effect defined as the percent change in the (average) level of the observed endpoint during a chronic external $\gamma$ irradiation exposure experiment, named $EDR_{50}$ expressed in $\mu$Gy/h (Garnier-laplace et al., 2006). Their geometric mean $HDR_{50}$ and associated 95% confidence interval can be easily calculated, still expressed in $\mu$Gy/h. Before being used to calculate the effect factor associated to a given radioactive substance, this working endpoint expressed as dose rates need to be converted into corresponding medium concentration (i.e. water and sediment for freshwaters). For a given radionuclide $r$, this conversion from dose rate endpoint ($HDR_{50}$ in $\mu$Gy/h) to corresponding medium concentration ($HC_{50r}$) needs to implement (i) a transfer sub-model to take on board all potential exposure pathways (external and internal irradiation), (ii) a dosimetric sub-model to calculate the energy absorbed by the organism from each radionuclide sources: water, sediment and the organism itself. The relationship between the activity concentration of an organism or media and internal or external absorbed dose rates is described by the dose conversion coefficient ($DCC; \mu$Gy/h per Bq/kg fresh weight). The method used to derive the $DCC$ values described by Beaugelin et al. (2006) has been applied to a suite of reference organisms representative of the variety of
freshwater biota in terms of mode of life and geometry to screen all ecologically relevant exposure scenarios.

The equations and associated parameters needed are specific to each (radionuclide \( r \), organism \( o \)) as follows:

\[
EC_{50}^{w,r} = \left( \frac{HDR_{50}}{DCC_{ext}^{w,r,o} + BCF_{w,r,o} \times DCC_{int}^{w,r,o}} \right) \quad \text{and} \quad HC_{50}^{w,r}
\]

corresponding to the geometric mean of all \( EC_{50}^{w,r} \) from the suite of reference organisms;

with: \( DCC_{ext}^{w,r,o} \) and \( DCC_{int}^{w,r,o} \), the dose coefficient to apply to the considered radionuclide for an external and internal exposure of the organism \( o \) to water respectively (\( \mu \text{Gy/h per Bq/L} \) or per Bq/kg), \( BCF_{w,r,o} \), the bioconcentration factor related to the aggregated transfer of the radionuclide from water to the organism \( o \), including the trophic pathway (L/kg fresh weight).

Uncertainties on both fate and effect factors were propagated along the calculation. The relative ranking of stressors is then done on the basis of the change in PAF per year due the annual releases.

**MAIN RESULTS AND DISCUSSION**

The method has been applied to liquid releases from nuclear power facilities along the Rhône river watershed, under normal operating conditions. These releases are characterized by chemical and radioactive mixtures and by yearly based emission rates. The chemical and radiological composition of low level radioactive liquid effluents produced by NPPs in 2005 (EdF, 2005) is reported on Figure 1. The difference between the sites in the relative contribution of each substance, stable or radioactive, is due to the variety of the process for operation and maintenance that can be site-specific. The Rhône river annual flow rate is given for the homogeneous section immediately downstream the emission point. The method enables to compare for 2005: (1) the relative importance of released chemicals and radioactive substances in terms of ecotoxicological impact (chemicals>>>radionuclides); (2) the major contributor in terms of impact for each category (chemicals: Cu and radionuclides: C-14) and (3) the relative contribution of different nuclear sites at the watershed scale (for chemical impact: St Alban>Crusas>Tricastin>Bugey) For any nuclear power plant along the Rhône river, the total percentage of potentially affected species for freshwaters remains lower than 1%. The PAF variation in one year is \( 10^3 \) times to \( 10^5 \) times higher for chemicals (Cu>Zn>H\textsubscript{3}BO_{3}) than for radioactive substances (C-14>Ag-110m>Co-60) according to the site.
FIGURE 1. Contribution of chemical and radioactive releases to the change in PAF in one year (Rhône river and NPPs releases in 2005 according to data from EdF)

REFERENCES


