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Spatially explicit risk analysis: A new solution to contamination problems in the Metropolitan Delta

N.W. van den Brink and J.M. Baveco

Alterra Green World Research

Wageningen, The Netherlands

nico.vandenbrink@wur.nl

Introduction

The Metropolitan Delta in North West Europe (further referred to as the Metropolitan Delta) can generally be characterised as densely populated, with high claims on land use. Due to changes in demography, economic prosperity and time expenditure of people, there is an increasing demand for areas for nature conservation/development and for recreational use. Such areas should optimally be located nearby urbanised areas in order to create a better balance between urban development and the open countryside or nature areas. However, since space is scarce in the Metropolitan Delta, it is not always feasible to allocate areas which are optimal suitable to nature development or recreation. More often, nature development and enhancement of recreational functions will need to be achieved in conjunction with other functions, or in areas with currently low-quality functional land use, which may be upgraded. An example of multifunctional use is the development of natural areas in floodplains in The Netherlands and Belgium, which have a prime function in flood control (see for instance Grift, 2001). Examples of possible changes in land use are for instance the regeneration of brownfields, which used to be industrialised areas (see for instance De Sousa, 2003).

Although both solutions, multi-functionality and changes in land use, appear auspicious, there is a major drawback which may hamper the development of natural and recreational areas, namely the occurrence of contamination in the soils. Not only the former industrialised areas, like the brownfields, but also other areas within the Metropolitan Delta have been contaminated by human activities. For instance, in The Netherlands, Belgium and Germany large areas in the catchment area of the river Rhine and Scheldt are polluted by a wide range of chemicals, like heavy metals, PCBs and dioxins (Hendriks et al., 1995; Vandecasteele et al., 2003). Other patterns of contamination are located near for instance smelters where elevated levels of heavy metals can be found (Janssens et al., 2003; Nahmani & Lavelle, 2002) or municipal waste incinerators which may be sources of air-borne dioxins (Domingo et al., 2002).

In order to assess the risks that such contamination may pose to the successful development of natural areas, specific methods have been developed, based upon ecotoxicological knowledge, e.g. knowledge on the effects of contaminants on the functioning of organisms. Currently used methodologies are based upon state-of-the-art expertise on assessing ecotoxicological risks. Within these methodologies, the problems of contamination in spatial planning are defined in a scientific, ecotoxicological framework, which primarily results in a limited set of options when risks of contamination are present: either a change of the planned land use function (loss of prime objectives of the initial planning process) or remediation of the contamination (requires vast budgets and is as such mostly not feasible). Besides the fact that these risk assessment methodologies only result in a very limited set of options to solve the problems, they may even more be of limited use in spatial planning processes because they hardly acknowledge expertise from other disciplines that are relevant in spatial processes. For instance geographers, social or economical scientists or spatial planners are currently not directly involved in the process of risk assessment (lack of inter-disciplinarity) but neither are other relevant stakeholders in the planning process, like for instance farmers, local inhabitants, (local) authorities (lack of trans-disciplinarity). Such lack of inter- and trans-disciplinarity may limit the possibilities to reach a balanced solution (see Tress et al., 2003). Hence, currently the development of natural, but also recreational land use nearby urbanised areas within the Metropolitan Delta is hampered by the lack of proper tools to assess risks of contamination, that are inter- and trans-disciplinary, and that are focussed on feasible solutions, related to the *a priori* defined planning objectives.

In the current paper a new conceptual outline for the ecological risk assessment of contamination will be addressed, which will result in an increased inter and trans-disciplinarity of the process, and which is more focussed on reaching a solution in dealing with the contamination problem. This concept is focussed on a relatively small scale of planning, 10s of km.

Conceptual outline

To enhance the potential role of ecological risk assessment in the planning process, we need to extend the procedure in two ways. In the first place, risk assessment has to be spatially explicit, taking into account the spatial structure of the landscape (landscape ecotoxicology, see Johnson, 2002). Such a spatially explicit risk analysis of contamination includes the spatial distribution of contaminants within the area of interest, and combines this with spatially explicit uptake of the contaminants by organisms acting on different spatial scales, based upon their exploitation patterns of the habitat. A presupposition of the concept is that besides its presence also the habitat usage pattern of an organism is determined by the configuration of the landscape (see for instance Powell & Steidel, 2002; Bélisle et al., 2001).

In the second place, hooks are required that link the outcome of the risk assessment, for a given landscape, to habitat manipulation measures modifying the habitat exploitation patterns in a way that minimises contact between organisms and contamination, and so minimises risks of the contaminants. In the planning process, a landscape that does not meet acceptable risk criteria (standards) can be modified and evaluated again, in an iterative approach (figure 1). The set of potential manipulation measures depend on the state of the landscape, the outcome of the risk assessment, and on constraints and priorities set by the stakeholders.

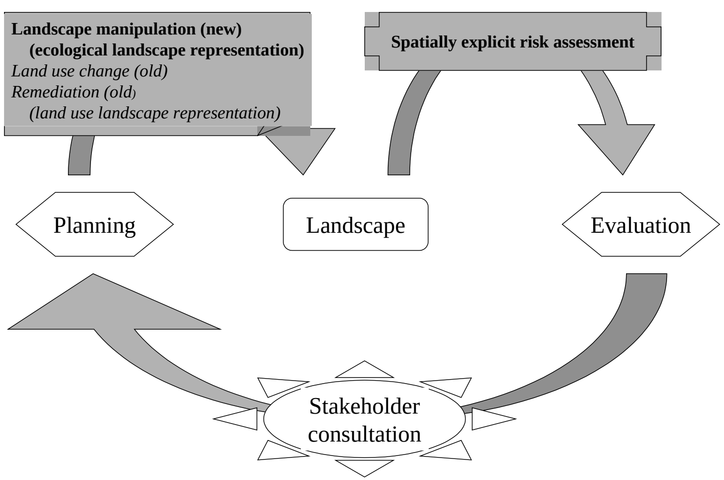


Figure 1. Schematic overview of the spatial planning process involving the spatially structured risk assessment (SSRA). The process of spatial planning results in some actions in order to minimise risks of contaminants of which landscape manipulation is only possible with the aid of a spatially explicit risk analysis. The resulting landscape plan can be evaluated with the SSRA, and when the risks are still unacceptable consultation with the stakeholders can result in alteration of the panning process.

In the a single top-predator species, or some selected species, with their food chain are selected for detailed analysis. Top-predators are chosen because they are located at the end of the food chain, they are in general prone to bio-accumulative contaminants like PCBs and cadmium and so vulnerable to risks of contamination, and furthermore they act at a similar spatial level as spatial planning processes. The selection of the species is based upon their place within the ecosystem, structurally and functionally, and when possible, it should be selected as such that it can be assumed that minimising the risks for the selected species, and its food chain results in minimising risks in general for the 'planned' ecosystem (Fleishmann et al., 2001).

The concept of the SSRA is relatively simple: it is aimed at minimising the contact between organisms and contaminants by spatially structuring the landscape so that the organisms will not forage at contaminated sites. For a SSRA, different types of information and tools are needed, which will be discussed in the following section. The SSRA can be applied at different phases in a decision process (Janssen, 1992). It can be applied in the development phase, aiding in optimising the design or it can be used in the selection phase, in order to evaluate choices. In an iterative process between stakeholders and scientists conducting the SSRA different scenarios can be assessed. Such a cyclic approach ensures the inter- and trans-disciplinarity of the process. This will be addressed later.

Information required for SSRA

When applying the concept to a certain case, information from different sources is needed. The planning process is the primary driving force in the concept. Based upon the outcome of this process a detailed analysis of the risks that contaminants may pose can be performed. In the following sections this will be illustrated by a hypothetical case.

The SSRA has to integrate information from different scales. Not only the scale of the planning is of concern, but also the specific characteristics of the selected species and the scale of the variability of the contamination patterns also influence the scale at which the SSRA may be applied. The example that is presented appears to be at a very small scale, but the concepts are also applicable at larger scales.

Information of the planning process

In a spatial planning process, a major input for the SSRA is a map with actual or hypothetical landscape configuration (figure 2). The map may reflect the outcome of the deliberations between stakeholders. These maps will contain the information on the habitat configuration that is planned, on the types of land use that are planned and so on. All this spatially explicit information is needed in the further steps of the process.

Information on spatial variation contaminants

In urbanised and rural areas contamination levels vary spatially. This may be due to several factors and processes. For instance, in river floodplains contamination levels vary with the rate of flooding. In areas with frequent flooding and sedimentation of particles the contamination levels are in general high (Middelkoop, 2000). In more urbanised areas the spatial patterns of contamination may be more related to human activities, like for instance dumpsites or contamination plumes in ground water due to a leakage. This spatial variation in contamination patterns can be mapped in a GIS driven system (Kooistra et al., 2001), and can be entered into the SSRA framework as a digital map (figure 3). The required level of spatial detail depends on the scale at which the planning process takes place, and on the degree of spatial variability of the contamination patterns. For further details on the mapping methods see Kooistra et al. (2001).

Information on spatial variation of organisms

The next step in the SSRA requires definition of habitat maps for the

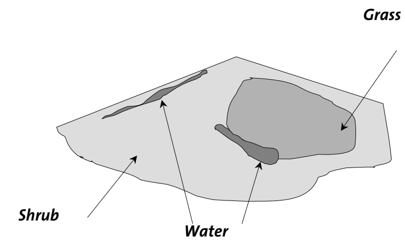


Figure 2. Example of possible outcome of a planning process in a hypothetical case-study in a certain area.

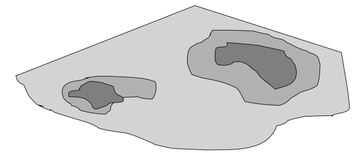


Figure 3. Example of possible spatial distribution of contamination in a hypothetical case-study in a certain area. The darker coloured areas contain higher levels of contamination.

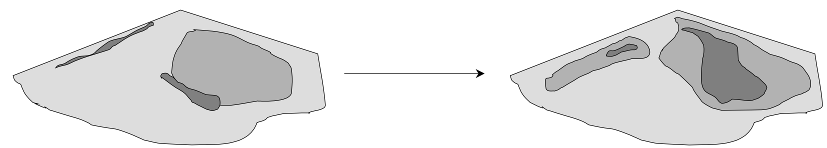


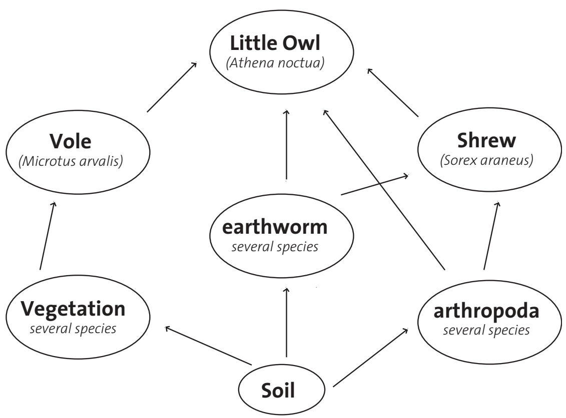
Figure 4. Example of a hypothetical derivation of a habitat suitability map from a vegetation map. It should be noted that this is very simplified, different types of information may be needed (for instance vegetation cover, groundwater level, depending on the species of interest)

species in the relevant food web. These species may operate at different spatial scales. At the smallest scale we may predict the presence of species, e.g. earthworms, and their population density, applying simple habitat suitability models and other (statistical) models relating a-biotic conditions to density (Morrison et al., 1992). At the larger spatial scale, referring to organisms more towards the end of the food chain, spatially explicit resource exploitation models are required, predicting which part of the specific landscape is exploited, and which food resource is consumed (figure 4). This information is essential, as it is part of the solution strategy to modify this spatial exploitation pattern, directly through manipulating food resource availability or indirectly by landscape changes affecting behaviour, e.g., removing shelter, etc.

Information on food-web relations

Contaminants are taken up by organisms through several routes. Uptake of for instance heavy metals by earthworms is dermal (Ma et al., 1998, Vijver et al., 2003), while uptake of contaminants in higher, terrestrial organisms, like for instance birds is mainly through dietary uptake (Drouillard 2000, Lovvorn & Gillingham, 1996). Depending on the food-web relations, different routes of contaminant uptake by predators can be distinguished. For instance for the little owl (*Athene noctua*), the food-web interactions in Dutch floodplains is depicted in figure 5 (Groen, 1997). Assessment of such food-web interactions is essential in order to quantify the uptake of contaminants by predators through food uptake.

Figure 5. Food-web interactions of the little owl (Athene noctua) in Dutch floodplains (cf. Groen, 1997)



Instruments required for SSRA

Models

Different types of models will be used in a SSRA. In this section we will not provide a detailed technical picture of the models, but a brief overview with possibilities and limitations of the use of these models will be given. Three types of models will be used in the SSRA: (i) models describing and analysing the spatial variability of the contaminants (spatial interpolation models), (ii) models used to construct habitat suitability maps for the species of concern (habitat suitability models) and (iii) models that address the transfer and accumulation of contaminants through the food-web (bio-accumulation models). The use of these models allows for the assessment of risks in hypothetical and realistic cases. In the planning process, many different potential situations are created (scenarios); being able to evaluate these scenarios is the crux of the SSRA. The models are used in a hierarchical fashion (each model building upon the output of a lower level model) and require high quality data-input (to avoid uncertainties to proliferate through the chain. Therefore, the models need to be validated in case studies, and furthermore each application in a certain case requires research effort in order to generate the data needed. Therefore the application of the instrument, and its ongoing development and fine-tuning may demand some research effort. Nevertheless, it may still be cost-effective because other solutions to deal with ecotoxicological risks e.g. remediation of sites or changes in planning priorities are even more costly or undesirable. Furthermore, those solutions generally do not incorporate stakeholder's participation.

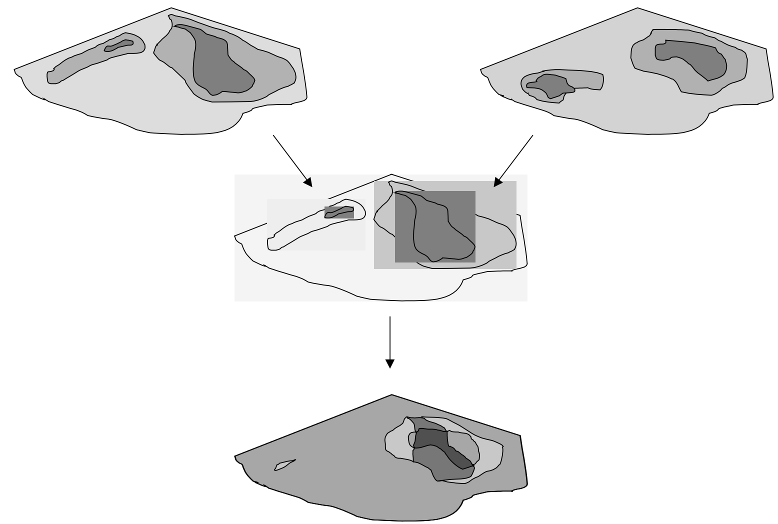
Decision support system (DSS)

In order to facilitate the iterative process between risk-assessors and other stakeholders a system is needed that consists of a framework that incorporates all models, and a graphical user interface through which the risk assessor and the other stakeholders can communicate with the models. Within the decision cycle such a decision support system (DSS) will assist during the planning phase to translate policy decisions into the actual implementation of measures, but it may also serve to evaluate established plans. It will enable to explore *a priori* several spatial planning alternatives, and to evaluate resulting effects on risks that contamination poses to wildlife. By applying the DSS the evaluation of the success of planned measures will take place early in the decision cycle, namely in the planning phase. Current methods only allow evaluation of risks after implementing of measure, e.g. monitoring of risks in the evaluation phase. Such an *a priori* evaluation of measures will result in a more effective implementation of planning measures and also allows that other main stakeholders are involved. When risks of contaminants are still to be expected in the newly defined configuration of the landscape, the consultation with the other stakeholders iterates once more, until a spatial configuration is defined that is not only acceptable to the stakeholders, but also with acceptable ecotoxicological risks. This will be addressed in the following section.

Assessment procedures

The procedure within the SSRA is an iterative cycle, in which the stakeholders supply the outcome of the planning process, formatted in maps, after which the scientific co-workers assess the risks. When the risks are unacceptable a new iteration will take place. The core of the risk assessment is relatively simple. From the maps containing the information on the habitat exploitation by the different species (based upon the maps of the planning results) and the spatial variability of the contaminants, a risk map is extracted by means of overlaying the contamination maps with the habitat exploitation maps (figure 6). For this overlay process GIS based algorithms can be used. This risk map is used to assess and value the risks of the contaminants.

Figure 6. Extraction of the risk-map as a combined overlay of the maps on the spatial variability of the contamination patterns and the habitat exploitation by the organisms.



Average Daily Intake (ADI) of contaminants by the species of concern can be calculated using the information on the contamination patterns, habitat exploitation of the organisms and the food-web models. Such ADI can be compared to known standards of ADI's at which no risks on effects are to be expected.

If the risks are acceptable, the SSRA can be concluded. If not, the SSRA allows for a cyclic approach in which the stakeholders can be consulted. The stakeholders can formulate news plans based upon their own needs and requirements, but combined with the information resulting from the risk assessment performed earlier (figure 1). This should lead to a change in the spatial configuration of the plans. How this planning process is managed is not of direct concern of the SSRA, only the outcome of the process is of importance. The maps of the new planning outcome can be entered in a new iteration of the SSRA and the risks of the contaminants can be assessed again. These iterations can take place until the risks of contamination are acceptable, and the resulting plans are thus the outcome of a combined effort of spatial planners and ecotoxicological risk assessors.

Relevance for the Metropolitan Delta

As said earlier the Metropolitan Delta can be characterised by high population densities, together with a high degree of urbanisation, which transforms rural and urban systems into large metropolitan areas connected by large infrastructure networks. Furthermore, agriculture systems have been intensified considerably in the last decades. These factors resulted in a deterioration of the availability of areas for natural development, but also for recreational use. This has been recognised at national level, but also at European level. For instance in The Netherlands it has been identified that nature development is fragmented, and considerable amount of areas should be devoted to nature development and the development of the Ecological Main Structure (RIVM, 2002). At European level this recognition has resulted in the adoption of the Natura2000 initiative. However, such recognition in policy plans does not automatically result in the fact that claims for space to realise these policy aims, will be complied with. Still, the interests of other stakeholder in the process need to be considered, so the spatial claims for nature development are likely not to be located at the optimal locations.

The approach that is presented here, needs a high degree of information input, e.g. data on spatial variation of contaminants, knowledge on habitat exploitation of organisms. Part of this information is generic available, but part will need to be collected case by case. This restricts the use of the concept to cases in which the collection of data is affordable, although it should be noted that for instance in The Netherlands information on contamination levels is legally needed by the owner transfer of land. Due to the intense character of planning processes in the Metropolitan Delta, and the high (financial) stakes that are involved, it is likely that the benefits of the concept in increasing the stakeholder participation and thus resulting in a better process in dealing with contamination problems, outweighs the extra efforts needed. This should also be viewed in relation to the fact that by applying these concepts it may be possible to redevelop contaminated areas that were formerly not dedicated to nature development or recreational use due to lack of proper risk-assessment tools.

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