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SPATIAL AND REGIONAL RISK ASSESSMENT IN DECISION SUPPORT SYSTEMS FOR ENVIRONMENTAL RISK MANAGEMENT

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SUMMARY

Environmental risks are traditionally assessed and presented in non spatial ways although the heterogeneity of the contaminants spatial distributions, the spatial positions and relations between receptors and stressors, as well as, the spatial distribution of the variables involved in the risk assessment, strongly influence exposure estimations and hence risks (Marinussen and Van der Zee 1996; Hope 2000; Korre et al. 2002; Linkov et al. 2002; Gaines et al. 2005; Makropoulos and Butler 2006). Taking into account spatial variability is increasingly being recognized as a further and essential step in sound exposure and risk assessment (Loos et al., 2006). According to the scale of the problem to be assessed, it is possible to identify two different approaches dealing with the spatial dimension of the risk assessment: the site-specific spatial risk assessment and the regional risk assessment. The first approach concerns the absolute risk assessment which is performed at local scale by the use of site-specific data collected through a characterization plan. The objective of this approach is the spatial estimation of the risks posed by some stressors (mainly contaminated sites) in order to support the allocation of the remediation alternatives and the definition of remediation plans (Carlon et al., 2008; Critto and Suter, 2009; Carlon et al 2009). At regional scale, the risk assessment deals with problems that affect large geographic areas where multiple habitats, sources, stressors and endpoints are present and their spatial relationship need to be evaluated (Hunsaker and al., 1990; Landis, 2005).

Due to the complexity of the spatial risk assessment process dealing with both, site-specific and regional scales, GIS tools, *ad hoc* spatial risk assessment methodologies and Decision Support Systems (DSS), involving several experts, stakeholders, and authorities, are required. Moreover, technical tools are needed in order to integrate the wide range of decisions related to contaminated land management and re-use, including environmental, technological and economic issues (CLARINET 2002).

In the present Ph.D. thesis the developed site-specific spatial risk assessment and regional risk assessment methodologies are described and their applications are presented.

As far as the site-specific risk assessment is concerned, the developed methodology applies geostatistic interpolation methods for mapping the distribution of contaminants concentration which are used in the risk characterization phase in order to provide a

zoning of the site based on the risk posed by multiple substances. Then, each identified risk-based area can be interrogated in order to provide information about most relevant contaminants of concern and exposure pathways. The final goal of the developed methodology is to support the formulation of remediation plans according to a stepwise spatial allocation of remediation interventions and an on-time simulation of risk reduction performances. The site-specific spatial risk assessment methodology was implemented in the DEcision Support sYstem for the REqualification of contaminated sites called DESYRE and applied to the Porto Marghera case study to support the entire remediation plans formulation process, encompassing hazard assessment, exposure assessment, risk characterization, uncertainty assessment and allocation of risk reduction measures.

Concerning the regional risk assessment, an innovative methodology which integrates a relative risk approach and spatial analysis was developed in order to select sites at regional scale where a preliminary soil investigation is required first. The regional risk assessment methodology was validated through the application to the case-study of the Upper Silesia region and was implemented within the Spatial decision support sYstem for Regional rIsk Assessment of DEgraded land, SYRIADE, which was developed by Consorzio Venezia Ricerche in collaboration with the European Commission JRC. SYRIADE is intended to be an aid for national and regional authorities in the inventory and assessment of (potentially) contaminated sites and mining waste sites at regional scale.

With reference to both, the regional risk assessment methodology and the related DSS, they resulted to be flexible tools which can be easily adapted to different regional contexts, allowing the user to introduce the regional relevant parameters identified on the basis of user expertise and regional data availability. Moreover, the GIS functionalities, integrated with mathematical approaches, allows to take into consideration all in once the multiplicity of sources and impacted receptors within the regional territory and therefore to assess the risks posed by all the contaminated sites in the region, thus supporting the final ranking objective.

SOMMARIO

Tradizionalmente nella valutazione dei rischi per l'uomo e per l'ambiente, la distribuzione spaziale della contaminazione, le relazioni spaziali tra le componenti dell'analisi di rischio e la distribuzione spaziale delle variabili coinvolte nell'analisi non vengono adeguatamente considerate, sebbene esse influiscano sulla valutazione dell'esposizione e quindi del rischio (Marinussen and Van der Zee 1996; Hope 2000; Korre et al. 2002; Linkov et al. 2002; Gaines et al. 2005; Makropoulos and Butler 2006). La necessità di includere la componente spaziale all'interno delle procedure di analisi di rischio è stata riconosciuta come un passaggio fondamentale nello sviluppo di appropriate valutazione di rischio (Loos et al., 2006).

In base alla scala di analisi, si possono identificare due approcci di analisi di rischio spaziale: l'analisi di rischio spaziale sito-specifica e l'analisi di rischio regionale.

Il primo approccio riguarda la stima assoluta del rischio che viene valutata a scala locale attraverso l'utilizzo di dati sito-specifici raccolti sulla base di un appropriato piano di caratterizzazione. L'obiettivo di questo approccio è la stima dei rischi posti da determinati stressori (principalmente siti contaminati) al fine di supportare la pianificazione spaziale degli interventi di bonifica e la definizione di piani di riqualificazione (Carlon et al., 2008; Critto and Suter, 2009; Carlon et al 2009).

A scala regionale, l'analisi di rischio si occupa di problemi che coinvolgono ampie aree geografiche caratterizzate dalla presenza di una molteplicità di habitats, di endpoints, di sorgenti che rilasciano una molteplicità di stressori e dalla complessità delle relazioni spaziali che si vengono a definire tra tutte queste componenti. Relazioni che devono essere opportunamente analizzate per supportare la stima del rischio regionale (Hunsaker and al., 1990; Landis, 2005).

A causa della complessità delle procedure di analisi di rischio spaziale, sia a scala sito-specifica che a scala regionale, diventa necessario utilizzare degli strumenti GIS e sviluppare degli strumenti *ad hoc* come i Sistemi di Supporto alle Decisioni (o DSS, Decision Support Systems) che permettono di coinvolgere diversi esperti, portatori di interesse e autorità all'interno del processo di valutazione e gestione dei rischi. Questi sistemi sono inoltre richiesti per integrare l'ampio ambito decisionale legato alla gestione dei siti contaminati e il ri-utilizzo, incluse le problematiche legate agli aspetti tecnologici ed economici (CLARINET 2002).

Nella presente tesi di dottorato è stata sviluppata una procedura di analisi di rischio spaziale sito-specifica e una procedura per l'analisi di rischio regionale. La prima proceduta utilizza metodi di interpolazione spaziale per ottenere delle mappe di distribuzione della contaminazione al fine di supportare la zonizzazione del sito sulla base dei livelli di rischio. Successivamente, ogni area omogenea, identificata sulla base del rischio, può essere interrogata per fornire informazioni relative ai contaminanti più rilevanti e alle vie di esposizione che più incidono sul rischio complessivo. L'obiettivo finale della metodologia è di supportare gli esperti e i decisori nella formulazione dei piani di riqualificazione, nella localizzazione spaziale degli interventi di bonifica e nella simulazione della riduzione dei rischi sulla base delle performance delle tecnologie.

La metodologia di analisi di rischio spaziale sito-specifica è stata implementata all'interno di DESYRE (DEcision Support sYstem for the REqualification of contaminated sites) ed applicata al sito di Porto Marghera (Venezia) al fine di supportare l'intero processo di riqualificazione del sito che comprende la valutazione dei pericoli, la valutazione dell'esposizione, la caratterizzazione del rischio e la localizzazione spaziale degli interventi di bonifica.

A scala regionale è stata sviluppata una metodologia innovativa che integra un approccio di analisi di rischio relativo con l'analisi spaziale, per selezionare i siti dove le attività di caratterizzazione sono urgentemente richieste. Questa metodologia è stata validata attraverso la sua applicazione al caso di studio della regione dell'Upper Silesia (Polonia) ed è stata implementata all'interno del sistema SYRIADE (Spatial decision support sYstem for Regional rIsk Assessment of DEgraded land) sviluppato dal Consorzio Venezia Ricerche in collaborazione con il Centro Comune di Ricerca (European Commission JRC) di Ispra. SYRIADE ha l'obiettivo di supportare le autorità regionali e nazionali nella predisposizione dell'inventario dei siti potenzialmente contaminati e dei siti di miniera e nella valutazione dei rischi che essi pongono.

In riferimento sia alla metodologia di analisi di rischio regionale che al relativo DSS, essi risultano essere degli strumenti flessibili che possono essere facilmente adattati a differenti contesti regionali, permettendo agli utilizzatori di inserire i parametri rilevanti, a livello regionale, selezionati in base alla disponibilità dei dati regionali e all'esperienza degli esperti coinvolti. Inoltre, le funzionalità GIS integrate con metodologie matematiche permettono di valutare contemporaneamente le relazioni tra le sorgenti e i possibili recettori al fine di stimare i rischi posti da tutti i siti potenzialmente contaminati presenti nella regione e quindi supportare l'obiettivo finale, ovvero la classificazione dei siti potenzialmente contaminati.

LIST OF CONTRIBUTIONS

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CHAPTER 1
Introduction

1.1 Background

Soil and groundwater contamination from localized sources is a worldwide problem which is often related to current activities, industrial plants no longer in operation, past industrial accidents and improper municipal and industrial waste disposals. In the European context, earlier industrialization and poor management practices have left a legacy of thousands of contaminated sites in Europe. There are an estimated 3.5 million potentially contaminated sites in the whole Union, of which about 0.5 million are expected to be actually contaminated and in need of remediation. Contaminants may accumulate to such an extent that they hamper soil functions, pollute groundwater and surface water and thus threaten drinking water supplies and aquatic ecosystems (EC, 2006a;
EEA, 2007).

To discuss the assessment and management of the soil resource, a Soil Thematic Strategy has been established in 2003 (EC, 2006a, EC 2006d). The strategy, guided by the European Commission, involved the EU Member States, Candidate Countries, European Institutions, Networks of Regional and Local Authorities and a broad community of European-wide Stakeholder Organisations. An Advisory Forum and five Working Groups were set up in different and critical aspects of management: 1) Soil Erosion, 2) Organic matter and biodiversity, 3) Contamination and land management, 4) Soil monitoring systems and 5) Research, sealing and cross-cutting issues (Van-Camp et al., 2004a).

The Working Group (WG) on contamination and land management introduced the Risk-Based Land Management framework. This guiding principle underlines three important aspects that shall be ensured when dealing with assessment and management of contaminated land: fitness for use (i.e. to ensure safe use or reuse of the land, taking risk acceptable for the people concerned); protection of the environment (i.e. to prevent or reduce negative impact on natural surroundings and to conserve or enhance quality/quantity of resources; long-term care (i.e., solutions must remain appropriate in the future) (Van-Camp et al., 2004b).

The strategy works has lead to the drafting of a Proposal for a ‘Directive of the European Parliament and of the Council establishing a framework for the protection of soil and amending Directive, 2004/35/EC’ was presented by the Commission on the September 22nd, 2006 (EC, 2006c). The aim of the Directive is to ensure the protection of soil, based on the principles of preservation of soil functions, prevention of soil degradation, mitigation of its effects, restoration of degraded soils and integration into other sectoral policies by establishing a common framework and actions. The Directive includes the principle of Risk-Based Land Management, as proposed by the contamination WG.

Among the obligations of the Directive regarding soil contamination Member States will be required to identify the contaminated sites in their national territory and establish a National Remediation Strategy on the basis of a common definition and a common list of potentially polluting activities. These plans will be based on the inventory of contaminated sites, and on a sound and transparent prioritization of the sites to be remediated, including timeframes, targets and allocation of resources (EC, 2006a).

The set up of an inventory of contaminated sites encompasses three main phases: a preliminary survey, a preliminary site investigation, and a main site investigation (EC, 2006b). In the first phase, on the basis of available information, a preliminary survey has the aim of assessing whether potentially polluting activities have taken place and whether contamination can be expected. As a result of the preliminary survey, a site will, in most cases, be classified as potentially (suspected to be) contaminated or not contaminated.

This first stage, preliminary survey, will be followed by a second stage, the preliminary site investigations to confirm the actual presence or absence of contaminants on the identified potentially contaminated sites.

Based on the outcome of the preliminary site investigation, the sites can be identified where the concentration levels of dangerous substances are such that there may be sufficient reasons to believe that they pose a significant risk to human health or the environment. On these "suspected" sites, as a third stage; full site investigations, will have to be carried out, including a risk assessment, to conclude if there is indeed a serious risk to human health or the environment. If so, then the site will be classified as contaminated sites and introduced in the inventory (EC, 2006b).

Preliminary site investigations in all the national or regional "suspected" sites may be very costly. As it is not feasible to investigate all land within the EU in a short timeframe with limited resources, attention should be given primarily to those locations that have a high potential contamination. Once the list of potentially contaminated sites is available, some Member States carry out a prioritization of potentially contaminated sites according to relative (or simplified) risk assessment procedures to select sites where a preliminary soil investigation is required first (Van-Camp et al., 2004). These risk assessment procedures are performed at larger scales (i.e. at regional scale) and should take into account that over a regional area, impacts caused by multiple sources are multiplied on diverse targets through different pathways. Moreover, these risk assessment procedures adopt a qualitative (or semi-quantitative) approach to the assessment of site risks. They describe the three components of the risk assessment (i.e. source, pathway and receptor) in term of scores, in order to estimate relative risks, rather than absolute estimates of health/ecological impacts. Once the prioritization of the potentially contaminates sites has been developed, a full site-investigation and a site-specific risk assessment should be performed for those sites which have been considered more risky.

According to the above considerations, the role of risk assessment in the development of the inventory of contaminated sites has a lead and dual role: in the planning of the preliminary site investigations phase where a relative risk assessment is required in order to prioritize the potentially contaminates sites to be characterized first and in the full site investigations phase where a site specific risk assessment is performed in order to assess the risks to human health and the environment and to define the site-specific clean-up levels.

Along with the emerging recognition of the relevance of spatial aspects in the management of contaminated land, taking into account spatial variability is increasingly being recognized as a further and essential step in sound exposure and risk assessment (Loos et al., 2006, Van-Camp et al., 2004, Carlon et al., 2008). In literature, on the basis of the scale of the problem to be assessed, two different risk assessment approaches can be identified which deal with the spatial dimension of the problem: the regional risk assessment and the site-specific spatial risk assessment, The first approach deals with problems that affect large geographic areas where multiple habitats, sources, stressors and endpoints need to be assessed and their spatial relationships need to be taken into account (Hunsaker and al., 1990; Landis, 2005). The objective of the regional risk assessment approach is the prioritization of the risks and the development of a ranking of potentially contaminated sites in order to identify those potentially contaminated sites where preliminary site investigations is required first.

The second approach concerns the absolute risk assessment which is performed at local scale by the use of site-specific data collected through a investigation plan. The objective of this approach is the spatial estimation of the risks posed by some stressors (mainly contamination) in order to support the allocation of the remediation alternatives and the spatial allocation of remediation interventions (Carlon et al., 2008; Critto and Suter, 2009; Carlon et al 2009).

The development and implementation of the two spatial risk assessment approaches and the presentation and discussion of the related results aim at supporting the decision making process, whose main challenge is to find sustainable solutions by integrating different disciplinary knowledge and different expertise and multi-actors views (Siller et al., 2004; Kiker et al., 2005). In fact, the decision making process for assessing and managing contaminated sites is controversial and difficult because of its diverse aspects such as economic interest, environmental restoration, social acceptance, technological application, land planning and requires effective spatial tools for the assessment and management of available spatial data and information (Gatchett et al., 2008).

Moreover, when public authorities have to manage complex contamination issues, tools that facilitate their challenging task in a framework that efficiently provides ideas, best practices and searchable resources are of great benefit. Technical tools are therefore needed in order to integrate the wide range of decisions related to contaminated land management and re-use, including environmental, technological and economic issues (CLARINET 2002).

Decision support systems (DSSs) are proven to be an effective support for any kind of decision-making process including contaminated sites management (CLARINET 2002). Decision Support Systems can be generally defined as tools that can be used by decision makers in order to have a more structured analysis of a problem at hand and define possible options of intervention to solve the problem (Jensen et al 2002; Loucks 1995, Simonovic 1996; Salewicz and Nakayama 2003). Going into detail, DSSs allow the integration of different types of information, they can include integrative methodologies, such as cost-benefit analyses, that evaluate site management alternatives, they can also provide powerful functionalities for analysis, visualization, simulation and information storage that are essential to complex decision processes.

Moreover, DSSs can facilitate one of the most important aspects of contaminated sites management, which is communication (Agostini et al., 2008). Finally, when spatial assessment needs to be taken into account in the analysis, Geographical Information Systems (GIS) functionalities can be included in the DSS thus resulting in Spatial Decision Support System (SDSS) (Malczewski, 1999; Carlon et al., 2008).

To the purpose of contaminated sites management, the currently available systems address in different ways the several involved issues: supporting the human health risk assessment procedures, providing selection of the most suitable technology to deal with the contamination of concern, or facilitating the identification of the best management solution for the site requalification (CLARINET, 2002). Some of them can facilitate the achievement of a shared vision for the redevelopment of the site of interest between both experts and stakeholders (Pollard et al, 2004); for example, by including Multi-Criteria Decision Analysis (MCDA) in order to allow consideration of different stakeholders priorities and objectives (Linkov et al, 2004; Giove et al, 2006; Kiker et al., 2005) and also integration of different issues. Although some specific tools for human health risk assessment are included within the existing DSSs, relevant development should be made in order to implement specific support system for spatial risk assessment both at regional and site-specific scale.

For this reasons, in order to support the risk based inventory of contaminated sites at regional scale, a Spatial Decision Support System (SDSS) was designed and implemented by the EU JRC and Venice Research Consortium. The system is called SYRIADE (Spatial decision support sYstem for RegIonal Assessment of DEgraded land) and allows to rank potentially contaminated sites and mining waste sites at the regional level; to rank risk sources hazard and receptors vulnerability; to integrate risk and socio-economic perspectives for the definition of integrated management areas. SYRIADE is a GIS (Geographical Information System)-based and MCDA (Multi-criteria Decision Analysis)-based SDSS that implements regional and relative risk assessments to support the ranking of potentially contaminated sites where investigation activities are urgently required. It is based on an integrated assessment of contaminated sites and mining waste sites, with consideration of both physical and chemical risks, as well as socio-economic aspects. Through its ranking results, the system effectively supports the inventory of (potentially) contaminated sites at regional.

For the investigation of human health risk at site-specific scale, the Venice Research Consortium and the University of Venice developed a software called DESYRE (DEcision Support sYstem for the REqualification of contaminated sites). DESYRE is a GIS-based decision support system (SDSS) specifically developed to address the integrated management and remediation of contaminated megasites (i.e., large contaminated areas or impacted areas characterized by multiple site owners and multiple stakeholders). The DSS covers all the different aspects of the remediation process, in a transparent and consensus-based decision-making approach, and defines management options by expert elicitation and stakeholders involvement. DESYRE provides an integrated assessment of risk and socio-economic issues, including innovative spatial and probabilistic risk assessment methodologies and remediation technologies selection, that may well support the risk-based approach to the contaminated site rehabilitation.

This thesis aims at presenting the Ph.D. work carried out by the author in the development and application of the spatial risk assessment methodologies implemented within the two above mentioned Spatial Decision Support Systems. The general objectives of the Ph.D. thesis were the development and application of innovative methodologies both for site-specific risk assessment and for regional risk assessment which were implemented within the spatial risk assessment module of the DESYRE SDSS and the regional risk assessment module of the SYRIADE SDSS, respectively.

Specific objective of the Ph.D. thesis were:

- the definition of the state of art concerning the spatial risk assessment both at regional and at site-specific scale including the assessment of the methodological similarities and diversities between the two approaches, the consequentiality of the implementation, the available tools already developed as well as knowledge and applicative tools gaps;
- the implementation of the deterministic and probabilistic spatial risk assessment methodology included within the DESYRE’s risk assessment module;
- the application of the deterministic and probabilistic spatial risk assessment methodology to a selected case study for its validation;
- the development of a regional risk assessment methodology for the ranking of potentially contaminated sites at regional scale;

- the implementation of the developed regional risk assessment methodology within the SYRIADE SDSS;
- the application of the regional risk assessment methodology to a selected case study for its validation.

The structure of the thesis is outlined in the next paragraph.

1.2 Thesis structure

The present PhD thesis is composed of 7 chapters: two theoretical chapters, two methodological chapters, two methodologies application chapters and a final conclusion chapter as reported below:

Chapter 2 - Spatial and regional risk assessment – where the theoretical background of site-specific and regional risk assessment approaches is described.

Chapter 3 - Decision Support Systems (DSS) for environmental risk assessment -, where the main characteristics of a DSS for contaminated sites are presented and the Multi-Criteria Decision Analysis (MCDA) is briefly introduced.

Chapter 4 - Spatial risk assessment in DESYRE- where the results of the research activities concerning the spatial risk assessment methodology developed and implemented in the DESYRE DSS is presented.

Chapter 5 - Regional risk assessment implemented in SYRIADE – where the results of the research activities concerning the regional risk assessment methodology developed and implemented in the SYRIADE DSS is presented.

Chapter 6 – DESYRE application to Porto Marghera case study – where the results of the application of the spatial risk assessment methodology to the Porto Marghera case study are reported and discussed.

Chapter 7 – SYRIADE application to the Poland case study– where the results of the application of the regional risk assessment methodology to the Upper Silesia (Poland) case study are reported and discussed.

Chapter 8 -Conclusions-, where final considerations on main findings of the work and possible further investigations and recommendations are presented.

CHAPTER 2

Spatial and regional risk assessment

According to the EU Soil Thematic Strategy, Member States shall be required to identify the contaminated sites in their national territory and establish a National Remediation Strategy on the basis of a common definition and a common list of potentially polluting activities. The sites to be remediated will be prioritized in a sound and transparent way, in order to reduce soil contamination and the risk it entails (EC, 2006d).

The efforts required to accomplish the identification and prioritization of contaminates sites in Europe are multiple and considerable (EEA, 2005a). Indeed, in order to know whether or not a site is contaminated, soil and groundwater investigations are necessary. However, as it is not feasible to investigate all land within the EU in a short timeframe with limited resources, attention should be given primarily to those locations that have a high potential contamination. In this perspective, a definition of “potentially contaminated site” is desirable: a “potentially contaminated site” is a “site where an activity is or has been operated that may have caused soil contamination” (Van-Camp et al., 2004a). Maps, historical risky activities, historical archives, local knowledge, industrial permits and license records, administrative information, surveys of surface and groundwater quality and site visits may give indications that a site in a specific region may be contaminated.

Once the list of potentially contaminated sites is available, some Member States carry out a prioritization according to relative (or simplified) risk assessment procedures to select sites where a preliminary soil investigation is required first (Van-Camp et al., 2004a). For those potentially contaminated sites which are recognised to be more relevant in terms of potential risk for human health and ecosystems, the following step concerns the plan of a site-specific investigation which supports the development of a site-specific risk assessment. The objective of the site-specific risk assessment is the identification of the areas which are in need of remediation (Carlon et al. 2008).

Both regional risk assessment and site-specific risk assessment entail a very important aspect of regional management of contaminated or potentially contaminated sites: the spatial resolution. In fact, spatial analysis is an important component of risk assessment and risk-based management for contaminated sites, because the problems addressed are inherently spatial (Linkov et al, 2002; Hope, 2005), e.g. chemicals in the environment are rarely distributed in uniform concentrations, fate and transport of chemicals occur relative to time and space. Additionally, interactions of receptors such as humans, wildlife, and fish species within the environment occur in biased, heterogeneous ways, often directed by demographic or habitat preferences (Clifford et al, 1995; Johnson et al., 2009). Moreover, spatial understanding of stressors, targets, impacts and ecological processes is particularly crucial because jurisdiction, which conditions the management options, rarely, if ever, coincide with biologically or ecologically significant spatial units and because landscape-level processes influence population dynamics and human impacts (Andersen et al. 2004). Finally, the spatial distribution of multiple habitats, multiple sources, multiple stressors and multiple endpoints as well as the characteristics of the landscape, the spatial and temporal distribution of soil and hydrogeology characteristics, the environmental settings, the current and anticipated use of land and the socio-economic situation influence the risk estimate (Bien et al. 2004; Landis, 2005). To support the inclusion and evaluation of the spatial aspects of risk assessment, geographic information systems (GIS) play an important role in the management of spatial data and information, in the spatial data preprocessing and modeling and in the spatial data and information visualization (Johnson et al., 2008, Worboys & Duckham 2004).

In relation with the advances in computer hardware and software which intensified the application of quantitative spatial analysis techniques that would otherwise be extremely tiresome and time consuming, the assessment of the risks posed by both natural and antrophic stressors to human and ecological targets makes progressively use of spatial information (Korre, 1999a). According to the scale of the problem to be assessed, it is possible to identify two different approaches to deal with the spatial dimension in risk assessment: the site-specific spatial risk assessment and the regional risk assessment. The first approach concerns the absolute risk assessment which is performed at local scale by the use of site-specific data collected through a characterization plan (US-EPA, 1998). The objective of this approach is the spatial estimation of the risks posed by some stressors (mainly contaminants) in order to support the allocation of the remediation alternatives and the definition of remediation plans.

At regional scale, the risk assessment deals with problems that affect large geographic areas including spatial relationships between multiple habitats, sources, stressors and endpoints (Hunsaker and al., 1990; Landis, 2005). In this approach, along with the traditional concepts of ecological risk assessment, ranking models are used in order to estimate the relative probability that some environmental negative effects, caused by anthropological activity, can occur (Hunsaker et al., 1989; Hunsaker et al., 1990; Suter, 1990; Graham et al., 1991; Suter, 2006; Landis and Wiegers, 1997; Landis, 2005).
In the next paragraphs a detailed description of the site-specific spatial risk assessment and the regional risk assessment is reported encompassing some useful definitions, methodological approaches and applications.

2.1 Site- specific spatial risk assessment

2.1.1 Definitions

Site-specific spatial risk assessment: a methodology which combines quantitative risk assessment procedures and spatial distribution of stressors and receptors to produce an assessment of risks at local scale (i.e. site-specific assessment) which provides geographical risk maps which preserve the significant spatial dimension of the risk in order to facilitate the understanding and the communication of the risks (Gay and Korre, 2006, Pistocchi and Pennington, 2006).
Source: an entity or action that releases to the environment or imposes on the environment a chemical, physical, or biological stressor or stressors (US-EPA, 1998a). With regard to the contaminated sites, the source area is defined as the location if highest soil or groundwater concentration of chemicals of concern or the location releasing the chemicals of concern (ASTM, 1998). In order to estimate the spatial distribution of the sources and the stressors intensity over the area of concern, transport models and interpolation methodologies such as kriging can be used (Isaaks and Srivastava, 1989). As far as the contaminated sites are concern, in order to identify the spatial distribution of the contamination and therefore to quantify the dimensions of the source and the volumes of contaminated medium, Korre and colleagues (Korre, 1999a; 1999b, 2002; Gay and Korre, 2006) developed a methodology that combines statistical

and geostatistical analysis tools with GIS tools for the quantitative and spatial assessment of contamination sources.

Exposure pathways: the course of a chemical of concern takes from the source area to an exposed organism. It describes the mechanism by which an organism is exposed to a chemical. The exposure pathway includes a source of release, a point of exposure, a transport/exposure medium and an exposure route (ASTM, 1998).

The spatial characteristics of this risk assessment components are strictly dependent on the hydro-geological and morphological characteristics of the landscape, the environmental behavior of the different stressors (e.g. contaminants, invasion of new species, disease etc) as well as the spatial distribution of the targets. In fact, the spatial distribution of the sources, the spatial distribution of the hydro-geological and morphological soil parameters, the presence of natural and atrophic barriers and the spatial distribution of the climatic parameters should be considered within the fate and transport models which should be spatially resolved in order to take into account, among other things, the relative distance between the source and the receptors. The latter also influences the time frame which elapses between the release or the development of the stressor and the impacts on the surrounding targets. With regard to the fate and transport of contaminants in the environment, different model are available (i.e. Sesoil, AT123D, Modflow etc.) and can be implemented within a GIS environmental (Bien. et al. 2004).

Finally, the exposure to the stressors strictly depends on the spatial distribution of the characteristical exposure parameters and ecological sensibility.

Receptors: for human health risk assessment receptors are defined as persons that are o may be affected by a release; for ecological risk assessment receptor are defined as ecological resources that are to be protected at the site. Because of the variety of ecological resources that may be present, the choice of the ecological receptors relevant to the site is an important part of the problem formulation (ASTM, 1998). Assessing spatial risks, the spatial distribution of human and ecological receptors as well as the spatial distribution of the stressors, play a key role in the definition of the risks spatial distribution since the distance between the two components of the risk assessment influences the risk estimates as well as the spatial distribution of the receptors exposure parameters. The spatial risk assessment allows to simultaneously assess and visualize the risks affecting different receptors characterized by their own characteristic exposure parameters. It has to be underlined that in the analyzed works (Korre et al. 2002; Bien. et al. 2004; Gay and Korre, 2006; Chen et al., 1998; Carlon et al. 2007; Morra et al. 2006; SADA; NORISC), the risk spatial distribution depends only on the contaminant spatial distribution while the receptors type and associated exposure remain constant.

Exposure assessment: exposure is defined as the contact of an organism (humans in the case of health risk assessment) with a chemical or physical agent (US-EPA, 1998a; US-EPA, 1998b). The magnitude of exposure is determined by measuring or estimating the amount of an agent available at the exchange boundaries (i.e., the lungs, gut, skin) during a specified time period. In order to estimate (qualitative or quantitative) the magnitude of the exposure, the magnitude, frequency, duration, and route of exposure should be assessed considering past, present, and future exposure scenarios. Estimates of current exposures can be based on measurements or models of existing conditions, those of future exposures can be based on models of future conditions, and those of past exposures can be based on measured or modeled past concentrations or measured chemical concentrations in tissues (US-EPA 1998b). As for the other variables involved in the risk assessment, those influencing the exposure assessment can have a spatial distribution over the area of concern. For human health risk assessment, the frequency, duration and route of exposure depend on the land uses carried out in the different areas of the site of concern, while the magnitude of the exposure is strongly influences by the spatial distribution of the contaminants concentration and the actives exposure pathways. The final objective of the exposure assessment is to estimate the type and magnitude of exposures to the chemicals of potential concern that are present at or migrating from a contaminated site. The results of the spatial exposure assessment, reported in suitable maps, are combined with chemical-specific toxicity information to characterize potential risks.

2.1.2 Motivation of the development of the site specific spatial risk assessment

Recent policy developments in Europe and North America are moving towards regulations which encourage a risk-based approach to contaminated land assessment (EC, 2006d; EC, 2000). A tiered approach is applied which initially uses simple generic models for screening purpose and subsequently, if necessary, more complex site-specific models can be applied in order to deeply investigate the environmental phenomena (Gay and Korre, 2006). Despite the wide application and improvements in risk assessment methodologies, some enhancements can still be made particularly concerning the spatial assessment of the land contamination phenomena (Gay and Korre, 2006). In fact, site-specific risk assessment procedure traditionally uses a single point value for all the variables involved in the estimation of risk caused by the assessed stressor. This produces a single risk value which is representative of the entire analyzed site without considering the spatial variability of the involved parameters over the site. Thus, traditionally, environmental risks have been assessed and presented in non spatial ways although spatial dimensions in risk assessment cover different aspects which should be considered. Moreover, many scientists (Marinussen and Van der Zee 1996; Hope 2000; Korre et al. 2002; Linkov et al. 2002; Gaines et al. 2005; Makropoulos and Butler 2006, Carlon et al, 2008) generally acknowledge that exposure and hence risk assessments are strongly influenced by the spatial positions of both receptors and stressors and by the heterogeneity of contaminant distributions and other environmental characteristics. For all the above mentioned reasons, taking into account spatial variability is increasingly being recognized as a further and essential step in a sound exposure and risk assessment (Loos et al., 2006) and the site-specific spatial risk assessment is therefore growing interest. This methodology which combines quantitative risk assessment procedures and spatial distribution of stressors and receptors to produce an assessment of risks at local scale (i.e. site-specific assessment) can provide geographical risk maps which preserve the significant spatial dimension of the risk and support the understanding and the communication of the risks (Gay and Korre, 2006, Loos et al., 2006).

In particular, site-specific spatial risk assessment can take into account the following aspects:

- the stressors' spatial distribution which can be obtain by the use of spatial interpolation methods;
- the spatial distribution of receptors and exposure parameters associated with the current and future land use of the site and the surrounding areas;
- the spatial distribution of the exposure routes which depend on the morphological and hydrogeological characteristics of the area of concern as well as on the environmental behavior of the investigated stressors;

- the application of spatially-resolved fate and transport models.

The result of the assessment is the spatial representation of the risk estimates and the identification of risk-based homogeneous areas in order to support the allocation of remediation actions and the definition of management objectives.

As far as the contaminated sites are concerned, the prediction of the spatial distribution of the contamination and the relative spatial positions of receptors can strongly influence estimates of exposure and hence of risk (Hope, 2000). In fact, stressors and receptors are located or distributed in space and exposure pathways are spatially distributed resulting from the spatial propagation of stressors (e.g. harmful substances) (Bien et al., 2004). Moreover, very often, the contamination is identified in small areas with high concentration (i.e. hot-spot: a local, restricted area where the concentration of one (or more) contaminant(s) is very high) and it is not appropriate to assume that the entire population is exposed to a mean contamination value corresponding to the high level concentration of the hot-spot area. In fact, in this context, the estimated receptor' exposure can lead to an unrealistic estimation of the risks (Carlon et al. 2004). Therefore, the spatial distribution of contaminants can support the identification of areas of the site which are characterized by different level of contamination and thus different level of risk. However, in the spatial risk estimation, in addition to the contaminants spatial distribution another important issue to be faced is the spatial estimation of the chemical fate and transport. Traditionally, the assessment of chemical fate and transport has been done using non-spatial multimedia models developed as "boxes" which include all relevant environmental compartments involved in the process. In the last decade, a great development has been observed in the field of chemical fate and transport modeling with the diffusion of models taking into account the spatial aspect of the phenomena (Pistocchi, 2005; Pistocchi and Pennington, 2006). However, multimedia assessment of pollutants pathways remains quite a complex issue and the prediction of spatial patterns of chemical concentration is a field of growing interest since it is still not completely understood and developed (Pistocchi and Pennington, 2006). For this reasons, when contamination samples are available, the definition of the spatial distribution of the stressors is obtained by the use of contamination distribution maps obtained throughout interpolation methods like IDW (inverse distances weighted) and geostatistic (such as Kiging, Isaaks and Srivastava, 1989) rather than using chemical fate and transport modeling. Furthermore, the risk may increase or decrease depending on a wide range of factors such as nature of the contamination, soil and hydrogeological characteristics, environmental settings, current and anticipated use of land as well as socio-economic situations (Bien et al., 2004). All this factors have a spatial distribution and can differently influence the risks posed by contaminated sites.

Finally, the integration and assessment of the spatial resolved risk assessment variables leads to the spatial estimation of the risks which are reported in maps format describing the spatial distribution of the risks over the area of concern. These maps can usefully support the zoning of the contaminates sites according to risk levels which provide the basic information for the definition of spatial priorities in order to guide the selection and spatial allocation of remediation activities (Carlon et al., 2007).

Since the development and improvement of spatial risk assessment require the collection, analysis and management of a huge amount of spatial data, they have been allowed and enhanced by the use of GIS which provide a very powerful and highly flexible tool that increase the sophistication of the risk assessment methodology (Korre, 2002). GIS provide a useful tool for spatially represent geographic features and a spatial platform to collect, organize, analyze and model spatial data (Chen et al., 1998). Moreover, GIS became a useful tool to help decision-makers to get immediate visualization of the results of the risk assessment with respect to current and anticipated future land use (Bien et al., 2006). For these reasons, the site-specific spatial risk assessment is always implemented in GIS platforms which often are combined with Decision Rules which lead to the development of Decision Support System (DSS) for contaminated land management (Gay and Korre, 2006; Chen et al., 1998; Carlon, 2007, Morra ar al., 2006; Bien et al., 2004; NORISC, SADA; NOMIRACLE). A detailed description of the main elements of DSS is provided in Chapter 3.

2.1.3 Site-specific risk assessment applications and open issues

Along with the emerging recognition of the relevance of spatial aspects in the management of contaminated land, in the last few years some methodological proposals have been made to include the spatial aspects within the site-specific risk assessment. These methodologies are going to be presented in this Paragraph. Historically, a major impulse to the research in this field occurred after the nuclear accident of Chernobyl in

1986 for dealing with radioactive diffuse contamination (e.g. Yatsalo et al. 1997). Dealing with mining related problems, Korre et al. (2002) coupled advanced geostatistics and exposure assessment to describe the spatial distribution of human health risk associated with ingestion of lead contaminated soil. A recent paper (Gay and Korre, 2006) provided a detailed description of the same method applied to the identification of most sensitive residential areas exposed to diffuse contamination.
More specifically for local scale and soil contaminants, four software packages, HHRA-GIS (Morra et al. 2006), HIRET (Bien et al. 2004), NORISC (http://www.norisc.com/) and SADA (http://www.tiem.utk.edu/~sada/help/) respectively, have been developed to assess the spatial distribution of human health risk. In HHRA-GIS and HIRET, the human health risk assessment was implemented in a widespread Geographical Information System (GIS) platform to generate human health risk assessment maps as a function of soil contamination and land uses. NORISC (Network Oriented Risk assessment by In situ Screening of Contaminated sites) and SADA (Spatial Analysis and Decision Assistance) are more comprehensive GIS-based software packages that include modules for sampling, risk assessment and selection of remediation techniques. In HHRA-GIS the integration of the exposure and effect information is performed on a grid domain in order to generate maps of iso-dose and iso-risk. SADA and HIRET are based on the geostatistical interpolation (kriging) (Isaaks and Srivastava, 1989) of soil contaminant concentrations, while NORISC estimates the risk for each sampling points separately and then defines risk zones based on Voronoi geometry algorithms (Okabe et al., 1992). In all the above software packages, risk maps are provided in raster format. It should be noted that the raster format can support the visualization of risk indicators, while other information about the estimated risk (e.g., contribution of different pathways and contaminants, or most sensitive receptors) is not retained, unless it is separately mapped. However, none of HHRA-GIS, SADA, HIRET and NORISC software packages supports the calculation of the distribution of uncertainty While HHRA-GIS and HIRET do not provide an operational link between the risk assessment and the selection of remediation techniques, SADA and NORISC offer decision support modules for the selection of remediation techniques. However, these modules can be applied only to the peak concentrations, i.e. they were not designed for planning remediation interventions over large sites.
Finally, DESYRE (DEcision Support sYstem for the REqualification of contaminated sites) is a GIS-based Decision Support System (DSS) specifically developed to address

the integrated management and remediation of contaminated megasites (i.e. large contaminated areas or impacted areas characterized by multiple site owners and multiple stakeholders). DESYRE was structured into six modules integrated into a GIS software platform: five assessment modules (*Socio-economic, Characterisation, Risk Assessment, Technological Assessment, Residual Risk assessment*) and a *Decision module* (Carlon et al., 2008). In the risk assessment module, the DESYRE software implements a spatial risk assessment methodology which allows the development of risk spatial distribution raster maps for each selected contaminant of concern (Carlon et al; 2008). Subsequently, the risk distribution raster maps are subjected to a process of vector transformation in order to obtain a zoning of the site according to risk levels, where each zone is identified as a vector object. Indeed, in DESYRE, outcomes of spatial risk assessment (the risk zones) are the basis for the selection and allocation of remediation technologies and the creation of remediation plans. In fact, the system allows the user to assign one technology, or a chain of technologies for each risk zone, and successively to simulate, step by step, the effect of the technological application by visualizing the residual risk map. The detailed description of the spatial risk assessment methodology implemented in the DESYRE system is reported in Chapter 4.

In the described site-specific spatial risk assessment applications some open issued can be identified. First of all, in all the describe methodologies the spatial distribution of human health and ecological risks are estimated through the use of the spatial distribution of the contaminants concentration and do not account for the spatial distribution of the exposure and hydrogeological parameters as well as for the morphology of the area of concern. None of the described site-specific spatial risk assessment applications deal with the spatial distribution of hydrogeological and morphological parameters as well as the spatial distributions of the exposure and ecological sensitivity parameters, although these parameters are important in the spatial estimate of the risks.
Thus, some further research activity concerning the site-specific spatial risk assessment should be focused on the following topics:
development of fate and transport models which take into account the spatial distribution of the hydrogeological and morphological parameters and their integration within the spatial risk assessment;

development of risk assessment algorithms which deal with the spatial distributions of the exposure and ecological sensitivity parameters.

Finally, it is also important to focus on cumulative risk. It is realized that human and ecological receptors are not exposed to individual substances in a relatively homogeneous environment, but to toxic mixtures in a heterogeneous environment. Single substances entail spatially variable environmental concentrations and variation in the combinations of these substances only increases the spatial variability of exposure and risk. A spatially explicit approach to cumulative exposure assessment might therefore generate more accurate exposure and risk estimates (Loos et al., 2006).

In more recent years, the open issued listed above concerning spatial risk assessment have been tackled in the NoMiracle project, NOvel Methods for Integrated Risk Assessment of CumuLative stressors in Europe funded by the European Sixth Framework Program. The project is aimed at developing novel methods and tools to better evaluate chemical risks which include novel spatial- and receptor-oriented approaches for assessing the integrated exposure to multiple stressors. Within the project, the spatial issue is mainly considered at two different levels: the estimation of the predicted environmental concentration by the use of regional fate and transport models and the estimation of the exposure of higher terrestrial organisms to contamination by the use of spatial exposure models (http://nomiracle.jrc.it/default.aspx).

As far as the estimation of the predicted environmental concentration is concerned, a regional fate and transport model was developed with the aim of providing computationally simple evaluation of fate and transport of chemicals through European water, soil and atmosphere using averaged, climatological environmental parameters at the monthly time step. The model is expected to provide a reasonable picture of trends in spatial distribution of chemicals of given origin and properties across the continent. The main use of such model is for support in policy making at the continental scale. Also, if realistic emission patterns are provided, the model provides a general evaluation of the impact of a given chemical in spatially distributed terms at the continental level. In that context, the model will be used for the mapping of EU-level concentrations and human exposure from emission. The model is NOT for detailed assessment of the fate and transport of chemicals in the vicinity of emissions, where local effects unaccounted for are relevant (Pistocchi, 2005).

Within the NOMIRACLE project in order to estimate the exposure spatial distribution, a spatially explicit model supporting the exposure estimates of higher terrestrial organisms to contamination, was developed taking into account spatial variation in contaminant concentrations and habitat characteristics, and food-web relations. This model improves the understanding of complex and cumulative exposure situations and can be used to more accurately assess complex exposure situations in which the spatial positions of stressors and receptors are relevant (Loos et al., 2006).

In the procedure, an individual receptor moves over a multi-celled landscape, whereby it encounters and accumulates spatially variable amounts of contamination. Movement can either be random or governed by "rules of movement", representing a receptor-specific way of responding to variations in the landscape. Such an approach not only facilitates the incorporation of species-specific foraging behavior in a risk assessment procedure, but can also be applied to account for spatial variability in the duration of exposure or the presence of multiple contaminants and their respective spatial variability (Loos et al., 2006; Hope 2000, 2001, 2005).

2.2 Regional risk assessment

2.2.1 Definitions

Regional risk assessment aims at providing a quantitative and systematic way to estimate and compare the impacts of environmental problems that affect large geographic areas (Hunsaker et al., 1990).The regional risk assessment is defined as a risk assessment procedure which deals with spatial aspects and considers the presence of multiple habitats, multiple sources releasing a multiplicity of stressors impacting multiple endpoints as well as the characteristics of the landscape which affects the risk estimates (Landis, 2005). The main characteristic of the regional risk assessment is the complexity of the analysis caused by the presence of multiple sources releasing multiple stressors which can impact diverse receptors as well as by the regional scale of the analysis which implies the assessment and integration of a huge amount of input data.

Region: a spatially extended non homogeneous area which is defined on the basis of physical, industrial and economical characteristics and not necessarily on administrative boundaries. The region is not homogeneous in the sense that "*smaller spatial units exist within the region that are more homogeneous that the region*" (Hunsaker et al., 1990). Furthermore, the boundaries of the region depends on the dimension of the environmental problems to be assessed, on the potential areas that can be directly affected, on the physical or biological processes that affect the impact of the hazard, on the spatial characteristics of the regional landscape and on the strategic planning and management decisions scale (Graham et al. 1991, Smith at al. 2000; Gheorghe et al., 2000; Hunsaker et al., 1990; Suter, 1990).

<u>Source:</u> the cause of environmental hazard which impacts large areas (e.g acid deposition, nonpoint source pollution, increased global CO_2) or alternatively, multiple local factors which combined can create a regional hazard to a population, species or ecosystems (Hunsaker et al., 1990). Regional environmental problems involve risk sources that affect large areas, usually over long periods of time (Hunsaker et al., 1990).

<u>Disturbance/Hazard:</u> pollutant or activity and its disruptive influence on the ecosystem containing the endpoints (Hunsaker et al., 1989, 1990).

<u>Exposure pathways:</u> the course of a chemical of concern takes from the source area to an exposed organism. It describes the mechanism by which an organism is exposed to a chemical. The exposure pathway includes a source of release, a point of exposure, a transport/exposure medium and an exposure route (ASTM, 1998). In regional risk assessment, the spatiotemporal patterns of exposure to the hazard have to be estimated using transport models adapted to the regional scale as well as including in the assessment the spatial characteristics of the regional landscape and any other spatial characteristics which can be associated with the exposure (Graham et al. 1991; Hunsaker et al., 1990; Suter, 1990).

<u>Endpoints:</u> environmental entity of concern and the descriptor or quality of the entity (Hunsaker et al., 1990). In regional risk assessment, endpoints must be regional in scope, have emergent properties that exist at regional or landscape scale and their observation have to be made over large areas and long time period. To this end, remote sensing techniques are useful tools for collecting regional data on endpoints quality over long periods.

<u>Habitat:</u> the place in which an organism lives, which is characterized by its physical features or by its dominant plant types (Martin et al, 1990). Habitat incorporates all aspects of physical and chemical constituents along with the biotic interactions (US-EPA, 2002). In the regional relative risk assessment methodology proposed by Landis a

habitat is a group of receptors (Landis, 2005). The spatial distribution of habitat changes and availability over the region has to be investigated because it has important implications for population viability and regional biodiversity (Smith et al. 2000).
Regional relative risk model: a system of numerical ranks and weights factors developed in order to combine and assess different kinds of risks (Landis, 2005).
Reference environment: geographic location and temporal period for the risk assessment (Hunsaker et al., 1990).
Exposure habitat modification: intensity of chemical and physical exposures of an endpoints to a hazard (Hunsaker et al., 1990).

2.2.2 Motivation of the development of the regional risk assessment

The increasing number of contaminated sites present in the same region, the wide range of different types of sources releasing a variety of stressors which can impact a multiplicity of assessment endpoints and the presence of many environmental hazards which impact large geographical areas (e.g. acid deposition, non point-source pollution, increased global CO_2, acid rains, ozone depletion, global climate change, forest fragmentation, biodiversity loss, invasion of new species, etc.) require the development of risk assessment methodologies focused on regional scale (Suter, 1990; Hunsaker et al., 1990; Landis, 2005; Smith et al., 2000; Suter 2006). The aims of regional risk assessment methodologies are "*the description and estimation of the risks resulting from regional scale pollution and physical disturbance*" (Hunsaker et al., 1990). Regional risk assessment becomes important when policymakers are called to face problems caused by a multiplicity of sources of hazards, widely spread over a large area, which impact a multiplicity of endpoints of regional interest (Graham et al., 1991). In fact, both the Soil Thematic Strategy (EC 2006d) and the recent EU Mining Directive (2006/21/EC) (EC, 2006e) point out the need for the development of spatial risk-based methodologies for sustainable management of contaminated sites and waste mining sites at regional scale.
In comparison with the traditional risk assessment concepts, regional approach includes the spatial characteristics of the regional landscape and any spatial characteristics associated with the exposure or the effects of the exposure in the risk assessment (Graham et al. 1991). For the sake of example, the geographical distribution of soil and

groundwater degradation is influenced by geology, topography, climate, and on the uses of the land. These factors control the specific vulnerability to contaminants of soils and groundwater across Europe (Van-Camp et al., 2004). Moreover, the spatiotemporal patterns of exposures depend on the spatial relationship between the hazard sources and the endpoints and the spatially resolved fate and trasport pocesses. The spatial combination of these patterns influence the spatial distribution of the risks and many cumulative effects, which at local scale are not evident, can be apparent at regional scale.

In many cases, the limited economical resources do not always allow to plan remediation strategies that reduces all the identified risks to health, safety and environment identified at regional scale, and thus require the development of methodologies that rank risks in terms of their magnitude, rather than absolute estimates of health/ecological impacts. The main objective of this approach is the selection of those (potentially) contaminated sites which nedd to be investigated more thoroughly or to be prioritized for the planning of remediation actions (Long and Fischhoff, 2000). This risk-based approach is called relative risk assessment, since it provide a ranking of the more risky sites rather than an absolue estimation of the riks they pose. A review and analysis of the available relative risk assessment procedures for preliminary and simplified risk assessment of (potentially) contaminated sites was published in a report of the European Environment Agency (EEA, 2004) where 27 existing and documented international methodologies were analyzed. The reviewed methodologies are generally applied at the national or regional level for ranking (potentially) contaminated sites on the basis of available data in order to plan priority of actions in terms of detailed site investigation and, in some cases, direct remedial measures. All methodologies reviewed adopt a qualitative (or semi-quantitative) approach to the assessment of site risks. They describe the three components of the risk assessment paradigm (i.e. source, pathway and receptor) in term of scores, in order to estimate relative risks, rather than absolute estimates of health/ecological impacts.

The review allowed to identify and list the most common parameters used in the reviewed methodologies in order to support the development of a relative risk based methodology called PRA.MS: Preliminary Risk Assessment Model for the identification and assessment of problem areas for Soil contamination in Europe (EEA, 2005b).

PRA.MS uses a risk assessment conceptual model, which is based on the Source-Pathway-Receptor (S-P-R) paradigm, where the considered exposure routes are Groundwater, Surface Water, Air and Direct Contact for health risks, while Surface Water and Protected Areas are considered as receptors of ecological risks. The methodology follows a ‘nested’ architecture where, for every exposure route, some parameters are identified and grouped into factors, which are grouped into indicators. Indicators are integrated to provide a score value for every exposure route in the human health risk assessment, while for the ecological risk assessment, indicators are integrated to provide a score value for each receptor (EEA, 2005b).
However, none of the reviewed methodologies or the PRA.MS itself properly addresses the spatial relationships between the sources and the receptors. The development of a regional risk assessment procedure which integrates the relative risk approaches and spatial analysis in order to select sites at regional scale where a preliminary soil investigation is required first, is one of the objectives of this thesis. To this end, the PRA.MS procedure was further implemented in order to deal with problems that affect large geographic areas (regional scale), where multiple habitats, multiple sources, multiple stressors and multiple endpoints are present and connected by spatial relationships (Hunsaker and al., 1990; Landis, 2005). The integration between the relative risk assessment and the spatial analysis at regional scale led to the development of a regional risk assessment methodology which may be used by policymakers/decision makers when they are called to face with problems caused by a multiplicity of sources of hazards, widely spread over a large area, which impact a multiplicity of endpoints of regional interest. A detailed description of the developed regional risk assessment methodology is provided in Chapter 5.
It has to be underlined that the development of regional risk assessment approaches strongly depend on the availability of regional data and spatial data integration methods (Smith, 2000; Locantore et al. 2004) and has been supported by the used of GIS tools for spatial data management. Moreover, the major availability of remotely sensed data of the earth’s surface and environmental monitoring data offer enormous potential to assess regional changes in ecosystems. However, the huge amount of spatial data necessary for the regional risk assessment entails the development of Spatial Decision Support System that can integrate remotely sensed data, GIS tools, spatial data integration methods, models for forecast changes in ecosystem resiliency and predictive

modeling for prioritization issues and management actions (Patil et al. 2001; Smith, 2000).

2.2.3 Different approaches for regional risk assessment

In order to face with the problems concerning the development of regional risk assessments two different approaches have been developed and described in the following paragraphs. The first approach developed by Hunsaker and colleagues (Hunsaker et al., 1989; Hunsaker et al., 1990; Suter, 1990; Graham et al., 1991) uses the traditional concepts of ecological risk assessment except that they analyse exposure and responses over a large area (Suter, 2006). The second approach developed by Landis and colleagues (Landis and Wiegers, 1997; Landis, 2005) uses ranking models to estimate the relative probability that some environmental negative effects, caused by anthropological activity, can occur. Indeed, the difficulty of obtaining exposure and effects measurements for regional scale assessment has lead to the development of regional relative risk assessment methodologies which overcomes this problem assigning to each risk assessment component a relative risk score.
In the following paragraphs these two approaches are described.

2.2.3.1 The regional risk assessment approach (Hunsaker et al., 1989)

The regional risk approach proposed by Hunsaker and colleagues has been developed in order to provide a quantitative and systematic way to estimate and compare the impacts of environmental problems that affect large geographic areas (Hunsaker et al., 1989; Hunsaker et al., 1990; Suter, 1990; Graham et al., 1991). The proposed approach combines regional assessment methods and landscape ecology theory with an existing framework for ecological risk assessment (Hunsaker et al., 1990). While traditional concepts and methods in risk assessment are relevant mainly to single sites or small geographical areas and are developed to assess the effects of single stressors (i.e. a single industrial effluent) on a single endpoint, regional risk assessment concerns the evaluation of the impacts which occur at regional scale on population, species or ecosystem that are widely dispersed over a region. Moreover, regional risk assessment can be used to evaluate the impacts of multiple local factors which can be combined to create a regional hazard. The regional risk assessment includes 5 key steps (Hunsaker et al., 1990):

- qualitative and quantitative description of the source terms of the hazard (e.g. location and emission levels for pollutants sources);
- identification and description of the reference environment within which effects are expected;
- selection of endpoints;
- estimation of spatiotemporal patterns of exposure by using appropriate environmental transport models or available data and
- quantification of the relationship between exposure in the modified environment (reference environment) and effects on biota.

Regional risk assessment has two distinct phases: the definition phase and the solution phase as reported in Fig 2.1.

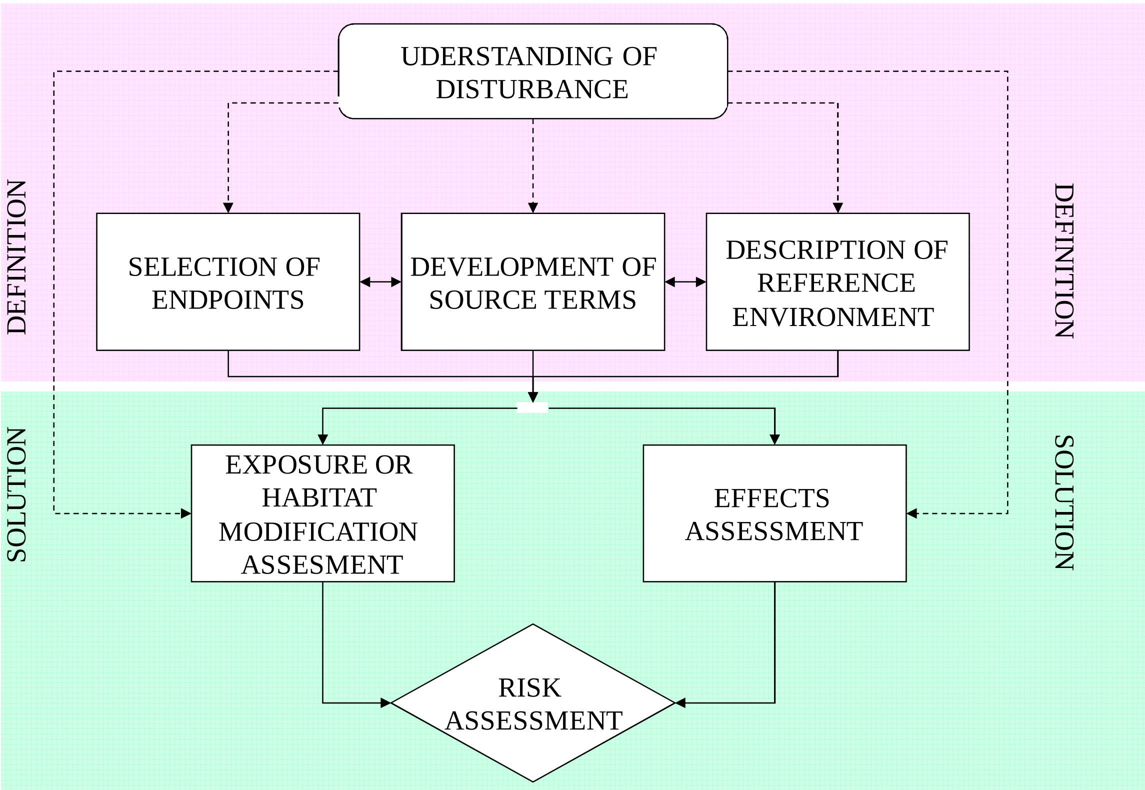

Figure 2.1. Regional risk assessment is composed of two distinct phases: the definition phase and the solution phase (Figure from Hunsaker et al., 1989).

The two phases can be found in local risk assessment as well as in regional risk assessment; nevertheless, some differences between the two approaches can be identified. Concerning the definition phase, in regional risk assessment, the concept of hazard is more nebulous and the interaction between the components of the definition phase are often complex because source terms, endpoints and reference environment are all interdependent. Indeed, developing source terms can be difficult for regional hazards because they often involve multiple sources that vary in both space and time. Moreover, in the selection of the endpoints, one must consider not only ecological processes but also pertinent social, economic and institutional processes of the reference environment, in this case at regional scale.
In the solution phase, regional assessment differs from local scale in two ways. First, the models used in the exposure and effect assessment must be regional: local models may have to be adapted to larger geographical regions or very different models developed. Second, the exposure or effects assessment must account for uncertainty that may arise because of spatial heterogeneity, a feature that may not be significant in local assessment.
To deal with regional risk assessment, probabilistic spatial model can support the risk quantification. Indeed, the regional risk approach under analysis has used Monte Carlo techniques to quantify the risks posed by a spatial distributed hazard by the use of different hazard scenarios. The spatial impacts of the hazards on some measurements endpoints have been estimated by the use of Monte Carlo iterations of a spatial simulation model. The spatial model has been built in order to simulate, on each cell of the raster map describing the spatial distribution of the quality of the endpoints, the changes on the endpoints quality caused by the hazard impacts. The comparison between the baseline endpoints quality and the simulated endpoints quality has been made for each Monte Carlo iteration in order to determine the fraction of the Monte Carlo model iterations in which the endpoints measure shifted by more than a defined percentage (e.g. 10% for detectable change and 25% for a significant change). The shifting percentage is estimated dividing the number of cells which have changed their endpoints quality by the total number of cells. Since the endpoints quality is represented by a qualitative attribute rather than a quantitative attribute, the shifted endpoints measure is immediately defined. Finally, the risk probability is defined by dividing the fraction of Monte Carlo iterations which have shifted by more than a defined percentage, by the total number of iterations. The comparison of the risks posed by the

different hazard scenarios can describe how the change in the hazard intensity will impact the assessment endpoints.

2.2.3.2 The regional relative risk assessment approach (Landis and Wiegers, 1997)

The regional risk assessment approach developed by Landis and colleagues (Landis and Wiegers, 1997; Landis, 2005) is based on the consideration that, at regional scale, the complexity of the ecosystems is much more evident and consistent because of the multiplicity of sources releasing stressors, the multiplicity of habitats where the receptors reside and the multiplicity of endpoints. Indeed, the EcoRA methods traditionally evaluate the interaction of three environmental components: stressors, receptors and responses occurring at a single contaminated site. The interaction between these three components is described throughout measurements of effects. However, at regional scale, the risk assessment requires additional considerations concerning the presence of multiple sources, multiple stressors, multiple receptors and a large number of interactions which increase combinatorially with the number of environmental components involved in the regional risk assessment: "*stressors are derived from diverse sources, receptors are associated with a variety of habitats and one impact can lead to additional impacts*" (Landis and Wiegers, 1997). In order to expand the traditional risk assessment to a regional risk assessment, some additional consideration concerning the scale of the analysis, the complexity of the environmental structure and the spatial characteristics of the three environmental components are required. In the regional risk assessment developed by Landis and colleagues "*the three regional components are analogous to the three traditional components, but the emphasis is on location and groups of stressors, receptors and effects*" (Landis, 2005). Since "*these grouping are usually too indistinct to obtain overall measurements of exposure and effects*" which are required in the traditional risk assessment, a comparative approach has been used. Indeed, the Landis and colleagues approach for a regional risk assessment is "*to evaluate the risk components at different locations in the region, rank the importance of these locations and combine this information to predict the relative risk among these areas*" (Landis and Wiegers, 1997; Landis 2005). The relative risk model (RRM) for regional risk assessment developed by Landis and colleagues is "*a* *system of numerical ranks and weightings factors to address the difficulties encountered when attempting to combine different kinds of risks*" (Landis, 2005). The relative risk approach can be summarised in the following steps:

- identification of the different sources and their locations in the region;
- identification of the impacted habitats and their locations in the region;
- identification of the possible impacts and their locations in the region;
- ranking the importance of the different components of the risk assessment (sources, habitats and impacts);
- spatial visualisation of the different components of the risk assessment to verify if they overlap;
- relative risk estimation.

The relative risk estimation is obtained throughout the integration of the importance of the three components of the risk assessment (source rank, habitat rank and impact rank) and the interaction between them (spatial overlapping) (M. Hamamé, 2002). Indeed, "*if one component does not interact* (does not overlap) *with one of the other two components, there is no risk* " (Landis and Wiegers, 1997; Landis, 2005).

The first step for the development of the relative risk model (RRM) for regional risk assessment concerns the rescaling of the three environmental components. Instead of focusing the attention on a specific stressor released into the environment, on the receptors that live in the environment and on the receptor responses to the stressors, the rescaling of the risk components allow to focus on the source releasing stressors, on the habitats where the receptors live and on the ecological impacts which are a group of receptor responses. At regional scale, sources and habitats are more relevant than stressors and receptors, as well as, the range of possible impacts are more important that receptor responses to the stressors, also because, at this scale, the information concerning these groupings are easier to obtain than exposure and effects measurements. Moreover, in the relative risk assessment, a score is assigned to each component of the analysis in order to define its relative importance. This approach allows to combine and compare a variety of distinctly different measurements (e.g. chemical concentration and occurrence of an invasive species) taken with distinctly different unit of measurement (Landis, 2005). Once sources, habitats and impacts have been identified, spatially located and ranked, the following step is to subdivide the region in sub-regions. Then, for each sub-region, a relative risk score is calculated by integrating the scores of the three environmental risk components and a risk-based ranking of the sub-regions is provided.

Going into details, the regional risk assessment proposed by Landis and colleagues (Landis, 2005) can be summarised into the following ten steps:

- make a list of the important management goals for the region;
- make maps representing the spatial distribution of potential sources and habitats relevant to the management goals;
- break the region into sub-regions based upon a combination of management goals, sources and habitats locations and possible pathways of exposure and fate and transport processes;
- make a general conceptual model that will be used in each sub-regions and that links sources of stressors, habitats and endpoints;

* decide on a ranking scheme to allow the calculation of relative risk to the assessment endpoints;
* calculate the relative risks;
* evaluate uncertainty and sensitivity analysis of the relative rankings;
* generate testable hypotheses for future field and laboratory investigation to reduce uncertainties and to confirm the risk rankings;
* test the hypotheses listed in step 8;
* communicate the results in a fashion that portrays the relative risks and uncertainty in a response to the management goals.

The integration and further implementation of the two described regional risk assessment approaches lead to the regional risk assessment methodology developed in the present Ph.D thesis and described in detailed in Chapter 5.

CHAPTER 3

Decision Support Systems (DSS) for environmental risk assessment

3.1 DSSs definitions and objectives

Many definitions have been proposed for Decision Support Systems (DSSs) applied in different management fields (Pereira and Quintana 2002). Decision Support Systems can be generally defined as tools that can be used by decision makers in order to have a more structured analysis of a problem at hand and define possible options of intervention to solve the problem (Jensen et al 2002; Loucks 1995, Simonovic 1996; Salewicz and Nakayama 2003, Agostini et al., 2008). As a general characterization, DSSs are computer technology solutions that can be used to support complex decision making and problem solving. Conventional DSSs consist of components for database management, powerful modeling functions and powerful (but simple) user interface designs. In order to carry on its functions, a generic DSS is composed of the following components (Jensen et al 2002; Loucks 1995, Simonovic 1996; Georgakakos 2004; Salewicz and Nakayama 2003, Agostini et al., 2008):

- database(s) and data retrieval system;
- analytical models or algorithms;
- spatial analysis, usually performed through GIS;
- graphic and visualization tools, through Graphic User Interface;
- simulation and optimisation models.

The common objective of all decision support systems, according to Loucks (1995), is to “provide timely information that supports human decision makers – at whatever level of decision making.” However, a DSS can be alternatively used in different manners, as an information tool, as a learning tool, as a communication tool and as a management tool (Lahmer, 2004, Agostini et al., 2008).

In order to handle different temporal and spatial scales, the majority of DSSs include GIS (Geographic Information System) tools. These specific DSSs are often referred to as Spatial Decision Support Systems (SDSSs) (Agostini et al., 2008, Dietrich et al, 2004, Malczewski, 1999; Carlon et al., 2008).

Moreover, some DSSs are Web-based, in order to reach as many users as possible and allow information integration and sharing among different users. In these cases, in addition to the abovementioned basic elements, the components for user application (such as the web server and the client browser) are also included (Dietrich et al, 2004; Agostini et al., 2008).

Finally, in order to support decision makers in solving complex environmental problems, many DSS implement Multicriteria Decision Analysis (MCDA) methodologies which offer the ability to integrate policy preferences with the judgments of technical experts (Figueira et al., 2005; Linkov et al., 2007, Giove et al., 2009). MCDA methods enable simultaneous consideration of stakeholder interests and technical evaluations, utilizing rigorous scientific methods to process technical information. MCDA is especially important in situations of significant uncertainty and data scarcity, such as management and restoration of contaminated sites. In Paragraph 3.3 a brief introduction to the background of MCDA, with particular attention to environmental DSS, will be reported.

3.2 Decision Suppot Systems (DSSs) for contaminated sites management

In the remediation and management of contaminated sites the integration of different disciplines, knowledge and decision-makers points of view are critical. Moreover, in order to efficiently cope with the problem of remediation and management of contaminated sites many issues must be considered by site managers and interested parties, such as (Van-Camp et al., 2004): (potentially) contaminated sites prioritization, risk estimation, reduction of risk, socio-economic impacts on the area, technical suitability and feasibility, time and cost perspectives, possible reuse options, stakeholders' points of view. For this reasons, the management of contaminated sites appears to be a complex problem which is further accentuated by the inclusion of spatial evaluation in the different phases of the assessment i.e. the preliminary survey, the (potentially) contaminated sites prioritizations, the preliminary site investigations, the full site investigations and the remediation planning. This complexity derives from the need to select and plan site investigation and remediation intervention by balancing the environmental concerns caused by the contamination with the socio-economic constraints and benefits, the technological limitations, the social acceptance of redevelopment alternatives and the active participation of concerned stakeholders (Agostini et al., 2008). The complexity of the management process for contaminated sites is defined by the many assessment and management questions to be answered and by the choice of many possible intervention activities on the site (Pollard et al., 2004).

Computer-based systems aid site managers and interested parties in gathering and integrating information, selecting and applying analytical procedures and defining management options (Shim et al 2002; Ascough et al. 2002; Jensen et al 2002). In order to help decision makers in this critical task, Decision Support Systems may be proposed, due to their ability of elaborating and evaluating different data sets, presenting results in understandable formats and providing a common platform for consensus-based decisions.

A significant benefit that a DSS can provide to contaminated sites managers is the possibility to have a structured analysis of the process, in the best case from the problem formulation to the reuse of the site. The decision process can be guided by the DSS in each or in the majority of its decisional steps, by providing related information, suitable tools to address problems and possible optimal solutions, by encouraging discussion of necessary tradeoffs (Agostini et al., 2008).

In fact, DSSs can guide the users in the two phases of the redevelopment process (i.e. assessment and remediation) and can provide the appropriate tools to the decisions that are posed in each of them (Sullivan et al., 1997).

In the assessment phase, risk assessment represents the core of the phase. A DSS may provide a platform that supports the user in the characterization of the environmental situation and in the automatic calculation of risks. The spatial functionalities that some systems include can be used to visualize, for example, the conceptual model of the site, the contaminants distribution, to assess the spatial relationship between the environmental components of the risk assessment (i.e. sources, pathways and receptors) and thus to estimate the risk extension and to prioritize the contaminated sites which first need further assessment (Semenzin, 2006).

In the case of selection of sampling or remediation technologies, a DSS may aid the user by offering several criteria for an accurate definition and evaluation of the effects of the choices in terms of performances, costs, wider environmental impacts and so on (Bonano et al., 2000; Khadam and Kaluarachchi, 2003; Khan et al., 2004). The decision makers are facilitated in their selection because the system can automatically show advantages and disadvantages of each option and may provide a ranking based on decision makers preferences (Semenzin, 2006). In the assessment phase, another critical question is the involvement of experts and stakeholders. A DSS can facilitate the achievement of a shared vision for the redevelopment of the site of interest between both experts and stakeholders (Pollard et al, 2004). For example, Multi-Criteria Decision Analysis (see paragraph 3.4 for more details) can be included into DSSs in order to allow consideration of different stakeholders priorities and objectives (Linkov et al, 2004; Giove et al, 2006) and also integration of different issues (Semenzin, 2006).

This last feature of the DSS allow the inclusion of expert judgments in the integration of available information and the stake-holders point of view in balancing environmental concerns and socio-economic constraints for the definition of alternative management options, including the prioritization of potentially contaminated site and the analysis of multiple redevelopment scenarios.

In general, the use of DSS can provide transparency and openness to the process, since all decisions can be traced back by the system and can be accurately justified. The use of DSSs by management authorities may avoid the risk that decisions are taken only in consideration of partial information or with a disagreement of preferences (Pollard et al., 2004; Giove et al., 2006; Semenzin, 2006).

However, some issues can be identified when developing and using DSSs: difficulty in gaining acceptability, limitations in providing highly integrated information without the rationale behind clearly explained, need to be continuously updated, necessity to have reliable input data and clear assumptions (Sullivan et al., 1997). Moreover, as a general feature, a DSS should create a balance between the complexity needed to address the wide range of site conditions, and easiness to use (Sullivan et al., 1997). This is not simple to be supplied, because the system should be complete and helpful but at the same time easy-to-use, flexible in site-specific evaluations, manageable in terms of data collection and input, reliable in the correctness of results (Agostini et al., 2008; Semenzin, 2006).

Finally, the general characteristics of DSS for contaminated sites strictly depends on the specific contaminated site process questions which need to be addressed. In fact, when designing and building a DSS, the choice of functionalities and software components strictly depends on the management objectives that the system has (Agostini et al., 2008; Semenzin, 2006). As a consequence, as reported in the following paragraph, there is a great variety of Decision Support Systems and tools developed to answer specific questions of the process of contaminated sites management.

3.3 Identification of available Decision Support Systems for contaminated sites management

As mentioned above, different Decision Support Systems and tools have been designed to answer to specific questions concerning contaminated site management. In this Paragraph, few of those Decision Support Systems for contaminated site management are briefly described in order to give examples of the objectives which drive the development of Decision Support Systems for contaminated site management.

DESYRE (DEcision Support sYstem for REhabilitation of contaminated sites) is designed particularly to manage large contaminated sites. DESYRE is a GIS-based software composed of six interconnected modules that provide site characterization, socio-economic analysis, risk assessment before and after the technologies selection, technological aspects and alternative remediation scenarios development (Pizzol et al., 2009).

The ERA-MANIA DSS was developed to support the site-specific phase of the Ecological Risk Assessment (ERA) for contaminated soils. In particular, it is based on the Triad approach (Rutgers and Den Besten, 2005), where the results provided by three Lines of Evidence (LoEs) (i.e., chemistry/bioavailability, ecology and ecotoxicology) are gathered and compared to support the assessment and evaluation of the ecosystem impairment caused by the stressor(s) of concern (Semenzin et al., 2009).

RAAS (Remedial Action Assessment System) is a decision support system designed to assist remediation professionals at each stage of the investigation and feasibility study process. RAAS is based on two main components: ReOpt (which provides descriptive information about technologies, contaminants, or regulations) and MEPAS (which is a human health risk model). RAAS has the objective to identify remedial technologies for the specific site conditions, remediation strategy, and cleanup objectives, and to estimate the effects of applying those technologies (http://www.osti.gov/bridge/servlets/purl/6263513-x0b8ug/6263513.PDF).

SADA (Spatial Analysis and Decision Assistance) is a software that incorporates tools from environmental assessment fields into an effective problem solving environment.

The capabilities of SADA can be used independently or collectively to address site specific concerns when characterizing a contaminated site, assessing risk, determining the location of future samples, and when designing remedial action (Purucker et al., 2009).

The REC system includes three tools developed to evaluate risk reduction, environmental merit and costs of remediation alternatives. The use of the tools can be modular, which means that the three tools may be used independently, but the main aim is the integration of the risk, environmental impact and cost aspects (Nijboer, 1998).

The NORISC Decision Support System basically guides the development of a methodology for investigating and assessing a contaminated site, in particular, for determining the pollution occurrence in soil and groundwater, as well as the risks involved and the potential site reuse (http://www.norisc.com/).

The Web-based WELCOME IMS (Integrated Management Strategy) is a step-wise approach to establish integrated risk based management plans for large contaminated sites, from the initial screening to the final definition of the remediation scenarios and long-term site management plan. In fact WELCOME differs from the other systems because it provides an operational framework within which different tools are proposed for application to address specific concerns (for example, risk assessment or scenarios creation), but the outputs of the different applications are not linked together and elaborated by the system (http://euwelcome.nl/kims/index.php).

SMARTe (Sustainable Management Approaches and Revitalization Tools – electronic) is a free, open source, web-based, decision support system to help revitalize communities and restore the environment. It is primarily intended to help bring potentially contaminated land back into productive use. It contains resources and analysis tools for all aspects of the revitalization process from definition of future land use and stakeholder involvement to economic analysis of financing, market costs and benefits, environmental issues and liability aspects. It is a holistic decision analysis system that integrates these aspects of revitalization while facilitating communication and discussion among all stakeholders (Vega et al., 2009).

3.4 Multi-Criteria Decision Analysis (MCDA) methods

Environmental decision problem are usually characterized by an high level of complexity. In such a context, the Multi-Criteria Decision Analysis (MCDA) represents an important and crucial step (Munda, 1994). The MCDA consists of one or more procedure to assist the decision maker(s) during the phases of the decision process, and taking into account possible sources of uncertainty and/or different utility functions (Giove et al., 2006). Some techniques rank options, some identify a single optimal alternative, some provide an incomplete ranking, and others differentiate between acceptable and unacceptable alternatives (Kiker et al., 2005). Multi Criteria Decision Analysis (MCDA) includes a large class of methods for the evaluation and ranking or selection of different alternatives that considers all the aspects of a decision problem involving many actors (Giove et al., 2009).

A structural platform common to almost all the decision problems includes the following items:

- the decision maker (DM). A conceptual figure, a single person, a group of persons or an entity in charge of finding the best solution for the problem under assessment;
- a set A of alternatives, in the finite case: $A = \{a_1, \dots, a_m\}$, beside whom the DM must chose the best solution;
- a countable family of criteria or attributes or parameters[1], $K = \{k_1, \dots, k_n\}$. These are aspects of the problem which the DM considers crucial and they also define the alternatives. Criteria can be organized into a hierarchical structure, i.e. a decision tree where the root is the objective function whose leaves are the first-level criteria, each of them split again into second-level criteria (sub-criteria), and so on till the last level, whose terminal leaves are the indicators (or the last level sub-criteria) formed by the available information (data or judgments);
- an objective or target function (to be optimized) used to score, and in case rank, alternatives, usually an aggregation function;
- the decision maker's preferences for the different evaluation of the criteria;
- an algorithmic tool designed to optimize the objective function, considering all the above information.

[1] These three words can be seen as synonyms and are used indifferently throughout this thesis.

Typically MCDA can be subdivided in three main categories (Vincke, 1992): MAUT/MAVT (Multi-Attribute Utility/Value Theory), Outranking and Interactive methods. However, as discussed by Giove et al. (2006) and reported by Linkov et al. (2004), the applications of MCDA to environmental problems are mainly based on MAUT/MAVT methods, where the attribute values of each alternative are aggregated by means of a suitable "utility" function (or "value" function) to obtain the score of the investigated alternatives. This approach is based on the hypothesis of rational and consistency decision-maker (Bridges et al., 2004), and implies the existence of both the value functions and a suitable aggregation operator. Many methods exist to define the value functions (Keeney and Raiffa, 1976), but their description is beyond the aim of this thesis. Also the aggregation operator needs to be carefully selected and, as discussed by Giove et al. (2006), the simplest and most widely used aggregation function in the MAUT context is the weighted averaging operator (WA), where all the criteria values are multiplied by a weight, i.e. a real number defined by mathematical methods (e.g. AHP; Ramanathan, 2001). Finally, as for all decision processes, MCDA methods can be classified as single or multiple person. The latter one, where a group of experts or decision makers is involved, belongs to the Group Decision Theory. In this case the MCDA algorithms have to include suitable consensus measures showing how much the group of decision makers agree or disagree about the alternative ranking (Carlsson et al., 1992).

CHAPTER 4

Spatial risk assessment in DESYRE

The spatial risk assessment methodology developed in the DESYRE project was described in Carlon et al, 2008. The next paragraphs report the methodological section of this paper.

4.1 Remediation planning of contaminated megasites

The contamination of soil and groundwater can pose a significant risk for human health and the environment and, therefore, can limit the use or re-use of the site. Hence, the remediation of contaminated sites has the double objective of reducing the human and environmental risk to acceptable levels and allowing the re-use and re-development of the site. Since remediation techniques can have a limited level of effectiveness and be dramatically expensive, risk mitigation strategies need to be shaped on spatial and temporal priorities, identification of most sensitive receptors and selection of most suitable technologies.

This is especially relevant in the case of contaminated “megasites”. The term “megasites” is used to indicate large (km^2 scale) contaminated areas or impacted areas, like industrial harbours, petrochemical districts and mining areas, characterized by unacceptable costs for complete clean-up (within currently used regulatory timeframes) due to political, economic, social or technical constraints. Megasites are also characterised by: having multiple owners and stakeholders; the need for an integrated risk-based approach at a regional scale (Wolfgang and ter Meer, 2003; Rijnaarts and Wolfgang, 2003; WELCOME, 2004).

The remediation of megasites requires the spatial planning of interventions with an effective definition of spatial and temporal priorities (remediation plans). For this purpose, the spatial distribution of risk must be estimated on the basis of the presence of multiple contamination sources and the distribution of human and ecological receptors over the megasite. The characterisation and distribution of risk over the site provides the basis for the identification of risk hotspots and for the selection and allocation of specific technological interventions, in order to enhance synergies among single interventions and optimize the overall efficiency in risk reduction.

Thus, alternative remediation plans should be developed and discussed with stakeholders. Alternatives should be compared on the basis of environmental benefits (the reduction of human health and ecological risk), economic and socio-economic benefits (following the re-use of the site), economic costs and environmental impacts of remediation works (Carlon et al., 2007; USEPA,1999).

In order to support the overal decision process, a dedicated Spatial Decision Support System (SDSS) (Malczewski, 1999) can play a major role. The invoked SDSS should support the development and comparison of alternative remediation scenarios, where risk mitigation is related to the technical feasibility and costs of remediation interventions, as well as economic and social benefits after the re-use of the site (Pollard et al., 2004).

4.1.1 Spatial risk assessment

The risk assessment for contaminated sites requires the characterisation of the potential adverse health effects associated with human and environmental exposure to soil and groundwater contaminants. It is a systematic process usually divided into four steps: 1) hazard assessment, 2) exposure assessment, 3) toxicity assessment and 4) risk characterisation (US National Research Council, 1983; USEPA 1989; ECB, 2003). With reference to these four phases, the spatial resolution of risk assessment leads to five main methodological challenges:

1. in the <u>hazard assessment</u>, the selection of chemicals of concern (CoC) has to take into account concentration levels as well as their spatial distribution;
2. in the <u>exposure assessment</u>, representative contaminant concentrations for the overall site have little significance, whereas contaminant's spatial distributions have to be considered in relation to potential receptors;
3. in the <u>risk characterisation</u>, the spatial distribution of risk estimations has to be described. This can be achieved by the extrapolation of risk indicators that can be mapped. These mapping methods should retain to the possible extent multiple information about the risk, like most relevant contaminants, pathways, receptors and effects, associated uncertainty, that can be useful for the spatial planning of remediation interventions;

4. in the uncertainty analysis, the spatial distribution of risk estimations should be complemented with the spatial distribution of the uncertainty, in order to identify those areas characterised by high risk and high uncertainty values and thus requiring a better investigation plan;
5. for the definition of remediation plans , the risk spatial distribution should support the spatial allocation of risk reduction measures.

4.1.2 The spatial risk assessment in the DESYRE SDSS

For the purpose of supporting the formulation and comparison of remediation plans for large contaminated sites, also called megasites, a SDSS software called DESYRE was developed (Carlon et al., 2007). In DESYRE, outcomes of spatial risk assessment are the basis for the selection and allocation of remediation technologies and the creation of remediation plans. In a final module, each remediation plan is characterised by indicators of risk reduction, socio-economic benefits and economic costs.

In the following paragraphs the spatial risk assessment methodology implemented in the DESYRE SDSS is presented. The adopted technical solutions are described and discussed in relation to the above mentioned challenges in the spatial risk assessment, with also reference to potential alternatives and improvements. For the sake of clarity, an example of application in a case study located within the Porto Marghera megasite (Venice, Italy) is also provided in Chapter 6.

4.2 Methods

4.2.1 Hazard assessment

The aim is the selection of a reasonably small group of substances, out of all those monitored at the site, which are expected to play the major contribution of risk and to which the risk analysis application can be restricted. Since megasites are characterised by a heterogeneous contamination from both the points of view of spatial distribution and multiple typologies of contaminants (up to hundreds of contaminants measured in one site), the selection of the contaminants of concern (CoC) is a necessary, but sensitive phase. Beside the conventional selection of contaminants exceeding regulatory concentration thresholds, when available, the most popular procedure for the further selection of CoC, named Toxicity-Concentration screen, is based on the scoring of contaminants according to their maximum observed concentration and toxicity properties (USEPA, 1989). In some cases, the frequency of analytical determination above the method detection limits (M.D.L.) is considered: e.g. exclusion of substances with frequency of determination below M.D.L. less than 5% (USEPA, 1989).

The Toxicity-Concentration screen does not consider the spatial variability of contaminants concentration and is suitable for the identification of CoC in small contaminated sites, or portions of larger sites. Similarly, for large sites the application of criteria based on the frequency of determination > M.D.L. can miss small portions of the site (e.g. < 5% of the overall site) with relevant contamination.

In DESYRE, the Concentration Toxicity screen was modified for application to large sites and complemented with other criteria. The proposed procedure implies the application of 3 criteria:

1. regulatory criteria: selection of contaminants exceeding the regulatory concentration threshold, when available (e.g. the acceptable concentration limits in soil and groundwater established by the Italian regulation);
2. ranking of contaminants (in soil and groundwater) by the Concentration Toxicity screen method applied for the maximum concentration, according to USEPA, 1989;
3. ranking of contaminants (in soil and groundwater) by the Concentration Toxicity screen method applied for the mean observed concentration.

The first criterion, i.e. the regulatory one, allows to exclude a large number of chemicals that are not likely to pose any significant adverse effect to human health and environment. The reliability and significance of this filter depends on the derivation methods of regulatory limits enforced in the specific region; however its application is largely conservative and may significantly reduce the number of substances for which toxicological properties have to be assessed (criteria 2 and 3).

The Concentration-Toxicity screen is applied in criteria 2 and 3, with reference to the maximum and the mean observed concentrations, respectively. The objective is to identify the chemicals in a particular medium that, based on concentration and toxicity, are most likely to significantly contribute to the risks for exposure scenarios involving that particular medium (USEPA, 1989). According to the EPA method, the proposed procedure implies the calculation of a hazard score (HS), for each chemical in each medium (soil and water), following Equation 4.1 for non carcinogenic substances and Equation 4.2 for carcinogenic substances.

$$HS = \frac{C}{T_{no\_carc}}$$ Equation 4.1

$$HS = C \cdot T_{carc}$$ Equation 4.2

where C is the contaminant representative concentration at the site in the analysed medium, T_{no_carc} is the Reference Dose (RfD, mg/kg-day) and T_{carc} is the Slope Factor ($[mg/kg\text{-}day]^{-1}$). RfD is an *estimate of a daily exposure level for the human population, including sensitive subpopulations, that is likely to be without an appreciable risk of deleterious effects during a portion of a lifetime* (USEPA, 1989).

The Slope Factor is defined as *a plausible upper-bound estimate of the probability of a response per unit intake of a chemical over a lifetime*. It assumes for carcinogens the absence of concentration thresholds below which no adverse effects are expected.

Chemical-specific hazard scores (HS_i) are summed to obtain the total hazard score (HS_{tot}) for all chemicals of potential concern in a medium. A separate HS_{tot} will be calculated for carcinogenic and non carcinogenic effects (USEPA, 1989). The ratio of HS_i to HS_{tot} can be approximately regarded as the relative contribution of that chemical to the overall risk at the site. Chemicals are ranked according to their HS_i/HS_{tot} ratio and excluded if the ratio is less than 1% (a lower percentage can be set).

In criteria 2, the representative concentration at the site, in the specific medium, is the maximum concentration. It allows the identification of contaminants that, due to high concentration peaks, might pose a relevant risk in some portion of the site.

In criteria 3, the representative concentration at the site, for each medium, is the mean concentration. It allows the identification of contaminants that might pose a relevant risk due to their spread presence at significant concentration over the all site.

The application of criteria 2 and 3 lead to the separate consideration of 8 ranking lists: 4 for each criteria, of which 2 for the soil and 2 for water, distinguished for carcinogenic and non carcinogenic substances, respectively. In practice, the partial redundancy of the 8 lists usually allow for a reduced number of chemicals.

4.2.2 Exposure assessment

The spatial distribution of the Contaminants of Concern (CoC) has been estimated by the use of geostatistical analysis allowing the incorporation of the spatial continuity into the estimation procedure and providing uncertainty indications (Isaaks and Srivastava, 1989). For each CoC, the geostatistical interpolation process leads to a spatial raster map of the contaminant concentration. The resolution of the raster map depends firstly upon the kriging interpolation grid, which is defined in accordance to basic geostatistical rules (Isaaks and Srivastava, 1989), but can be lowered (bigger pixels) at the user convenience in order to facilitate the interpretation of the map. Moreover, lower resolution can speed up the computer calculation for risk assessment, which is carried out at pixel level. Maps are produced for contaminants in soil and water.

Acceptable concentrations in soil and groundwater are calculated on the basis of calculated exposure for on site human receptors. As represented in the exposure diagram in Figure 4.1, the considered impacted media are contaminated soil and groundwater, and exposure routes are ingestion of soil and groundwater, inhalation of vapours and dust from soil, inhalation of vapours from groundwater, dermal contact with soil and groundwater.

A preliminary modelling of lateral transport of the contamination in the groundwater medium through the application of the Domenico equation (Domenico and Schwartz, 1998) and dilution factors in the surface water has been implemented in the DESYRE system.

This type of modelling was recognised to be over-simplified in the case of large contaminated sites, because groundwater direction is usually variable and multiple surface water receptors can be identified. On the other hand, more accurate modelling is possible with the application of external software models, e.g. ModFlow-2000 (Harbaugh et al., 2000).

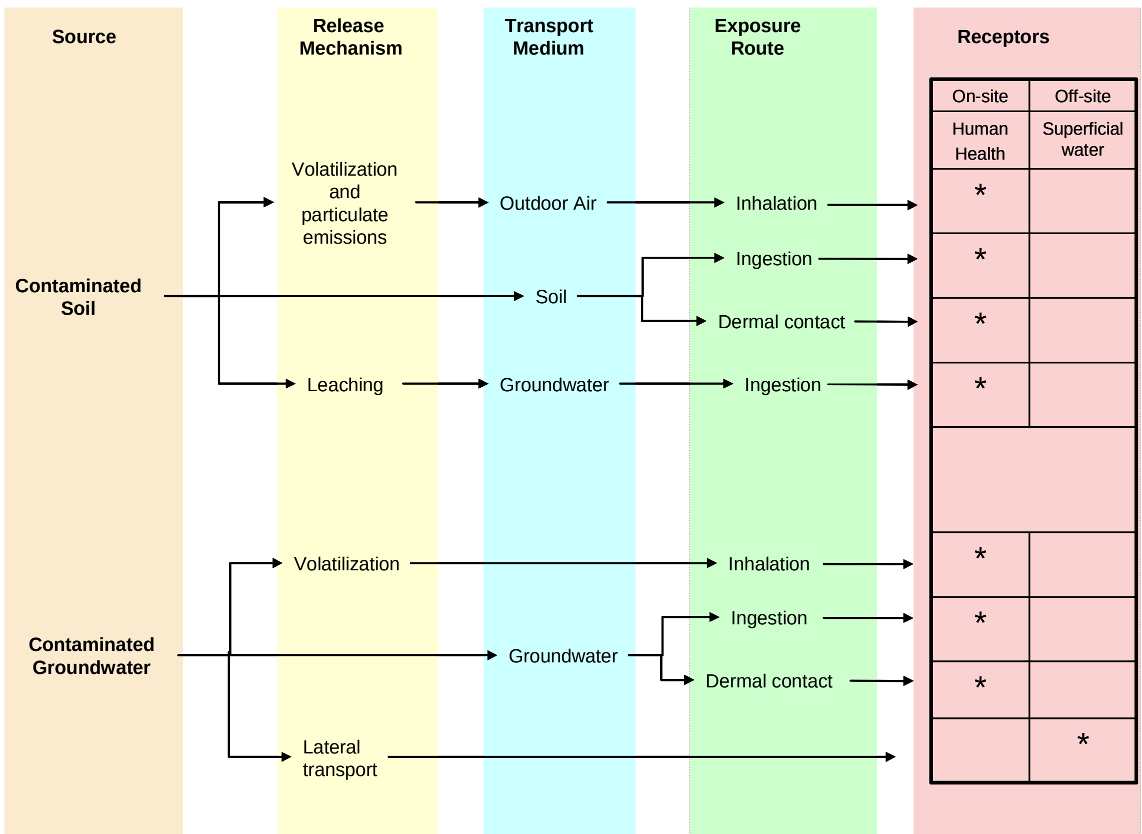

Figure 4.1. Exposure diagram that indicates contamination sources, release mechanisms, transports medium, exposure routes and receptors considered in the DESYRE software.

4.2.3 Risk characterization

4.2.3.1 Calculation of Risk Based Acceptable Concentrations

The software offers default exposure parameters for three scenarios: the residential, recreational and industrial land use, respectively. Exposure parameters can be changed by the user in order to fit better specific conditions of the overall site, or portions of the site.

In the DESYRE software the calculation of acceptable concentrations in the analysed medium is based on the standard algorithms elaborated by the American Society for Testing and Material (ASTM, 1998).

Concerning the toxicity assessment step, the DESYRE software is supplied with a toxicological database including Reference Dose for Ingestion, Inhalation and Dermal Contact and Slope Factor for Ingestion, Inhalation and Dermal Contact needed to characterise the risk. The selected toxicological values represent the more conservative values found in the following database: IRIS (USEPA, 2002) and RAIS (ORNL, 2002).

The multi-pathway acceptable concentrations in the soil medium and in the groundwater medium are calculated by integrating the acceptable concentrations calculated for the different exposure routes related to the specific medium according to Equations 4.3 and Equation 4.4, respectively (Norwegian Pollution Control Authority, 1999).

$$MACS = \frac{1}{\frac{1}{ACS_{ing}} + \frac{1}{ACS_{dc}} + \frac{1}{ACS_{inhal}} + \frac{1}{ACS_{ing-w}}} \qquad \text{Equation 4.3}$$

where:

$MACS$ = Multi - pathway Acceptable Concentration in the Soil medium

ACS_{ing} = Acceptable Concentration in the Soil medium related to the soil ingestion pathway

ACS_{dc} = Acceptable Concentration in the Soil medium related to the soil dermal contact pathway

ACS_{inhal} = Acceptable Concentration in the Soil medium related to the volatile compound and dust inhalation pathway

ACS_{ing-w} = Acceptable Concentration in the Soil medium related to leaching and water ingestion pathway

$$MACGw = \frac{1}{\frac{1}{ACGw_{ing}} + \frac{1}{ACGw_{dc}} + \frac{1}{ACGw_{inhal}}} \qquad \text{Equation 4.4}$$

Where:

$$\begin{aligned}
&MACGw = \text{Multi-pathway Acceptable Concentration in Goundwater}\\
&ACGw_{ing} = \text{Acceptable Concentration in Groundwater related to the water ingestion pathway}\\
&ACGw_{dc} = \text{Acceptable Concentration in Groundwater related to the water dermal contact}\\
&ACGw_{inhal} = \text{Acceptable Concentration in Groundwater related to the volatile compound inhalation pathway}
\end{aligned}$$

As mentioned above, a preliminary modelling of lateral transport of the contamination in the groundwater medium through the application of the Domenico equation (Domenico and Schwartz, 1998) and dilution factors in the surface water has been already implemented in the DESYRE system, although not yet tested. Therefore, its description is beyond the scope of this thesis.

4.2.3.2 Calculation of Risk Factor

The proposed spatial risk methodology is based on the estimation of the Risk Factor (RF) for a specific medium. RF is defined as the rate between the measured concentration and the acceptable concentration in that medium as expressed in Equation 4.5.

$$RF = \frac{ESC}{MACS} \qquad \text{Equation 4.5}$$

Where:

RF = Risk Factor;

ESC = Estimated Soil Concentration;

MACS = Multi-pathway Acceptable Concentration in Soil.

A similar equation is used to calculate the Risk Factor for groundwater: Estimated Groundwater Concentration (EGwC) and Multi-pathway Acceptable Concentration in Groundwater (MACGw) are used in place of ESC and MACS.

Based on the calculation for each cell of the raster concentration maps, the Risk Factor can be mapped for each contaminant. In order to obtain a more concise representation of the risk and to finalise the risk assessment results to the identification of suitable remediation interventions, the selected contaminants are divided into six categories, in accordance with the classification proposed by Federal Remediation Technologies Roundtable (FRTR, 2002): Nonhalogenated Volatile Organic Compounds, Halogenated Volatile Organic Compounds, Nonhalogenated Semivolatile Organic Compounds, Halogenated Semivolatile Organic Compounds, Fuels, Inorganics. These categories mainly depend on basic physico-chemical properties (e.g., water solubility, vapour pressure, bio-degradability) that heavily affect suitability and performance of various remediation interventions. In DESYRE the risk mapping is referred to contaminants categories, instead of individual substances. Each category includes a number of selected CoC. This allows to reduce significantly the number of risk maps and to make them more helpful for the formulation of remediation plans.

The Risk factor is estimated at each cell of the grid. For each category of contaminants, the Risk Factor can be referred either to the maximum value of RF (RFmax) or the sum of the values (RFsum) expressed by CoCs in that category. The selection of the maximum value is preferred when the aim of the analysis is to identify the risks caused by the hot spots, whereas the sum of the values represents potential cumulative effects of multiple contaminants within the same category. The comparison of the two maps (RFmax and Rfsum) supports the distinction between the case in which the risk is mainly associated to specific contaminants within a specific category from the case in which the risk is due to the cumulative contributions of several contaminants.

With respect to the conventional characterisation of risk in terms of Hazard Quotients and incremental Risk (UPEPA, 1989) for non carcinogenic and carcinogenic substances, respectively, the RF indicator has the advantage to provide a common scale for carcinogenic and non carcinogenic contaminants, allowing the comparison and summation of the risk posed by both of them.

4.2.3.3 Vector transformation

Raster maps of RFmax and RFsum represent the spatial distribution of the risk factor, but they do not provide a discrete zoning of the site and, more important, they do not retain relevant information about the risk characterisation, like the identification of most relevant contaminants, exposure pathways and impacted receptors, that is complementary to spatial features for the planning of interventions.

In DESYRE, a process of vector transformation overcomes this limitation. RFmax (or Rfsum) values are divided into five classes: $RF \leq 1$, $1 < RF \leq 3$, $3 < RF \leq 10$, $10 < RF \leq 100$, $RF > 100$, respectively. These risk factor classes were developed in order to support the decision maker in the evaluation and selection of the remediation technologies required for reaching an acceptable residual risk factor (i.e. RF = 1). The proposed RF thresholds values are referred to specific remediation technology performances which allow the achievement of a residual risk factor equal to 1 (i.e. 67% for RF equal to 3; 90% for RF equal to 10; 99% for RF equal to 100 and 100% for RF bigger than 100).

Accordingly, in the vector transformation process, all the cells, which are characterised by the same risk factor class and are concurrently spatially linked (i.e. adjacent), identify a homogeneous risk-based area. The transformation of the raster maps into vector maps leads to the zoning of the site according to risk levels, where each zone is identified as a vector object.

The vector transformation permits to link every zone (vector object) to a repository (i.e. a table) of intermediate results of the derivation of RFmax (or RFsum), such as the relative contribution of various contaminants in the same category and the relative contribution of various exposure pathways. This information can be popped up by clicking on one of the homogeneous areas and can support the expert to have a comprehensive understanding of the parameters that more influence the risk assessment results (e.g., Figure 6.1 for the case study).

4.2.4 Uncertainty analysis

The DESYRE software supports the probabilistic estimation of the risk factor, which also provides an indication of the propagation of the uncertainty in the input values into the risk estimate. The probabilistic risk assessment is based on the Monte Carlo analysis that is a popular statistical sampling technique for obtaining a probabilistic approximation to the solution of a mathematical equation or model. This technique is used to characterise the uncertainty and variability in risk estimates by repeatedly sampling the probability distributions of the risk equation inputs and using these inputs to calculate a range of risk values (USEPA, 2001). Within the DESYRE software, the Monte Carlo analysis has been used to calculate the value of the Acceptable Concentration in the source medium. For every selected substance and analysed pathway, this process leads to the definition of the empirical distribution of the acceptable concentration (AC) in the considered medium. The DESYRE software provides the possibility to introduce probability distributions for all input variables, encompassing all the exposure, physico-chemical, toxicological and hydrogeological parameters (e.g., Tables 6.2, 6.3, and 6.4 respectively in the case study application).

The parameters chosen to describe the Acceptable Concentration distribution are the 50th and 5th percentiles. The first one is related to the central tendency of the risk distribution, while the 5th percentile is related to the high-end of the risk distribution and is representative of the Reasonable Maximum Exposure (USEPA, 2001). The difference between the Central Tendency Exposure (CTE) and the Reasonable Maximum Exposure (RME) gives an initial indication of the degree of uncertainty in the risk estimate.

Similarly to Equation 3 for the calculation of soil MAC, in the probabilistic risk assessment the computation of the Multi-pathway Acceptable Concentration (MAC) in the analysed source medium was performed by the Equations 4.6 and 4.7. Equation 4.6 is used to calculate the 50th percentile (median) of the distribution of the Multi-pathway Acceptable Concentration (MAC), while Equation 4.7 calculates the 5th percentile. The source of the contamination is soil.

$$MACS(MEDIAN) = \frac{1}{\frac{1}{ACS_{ing\,MEDIAN}} + \frac{1}{ACS_{dc\,MEDIAN}} + \frac{1}{ACS_{inhal\,MEDIAN}} + \frac{1}{ACS_{ing\_w\,MEDIAN}}} \qquad \text{Eq 4.6}$$

where:

$\mathrm{MACS(MEDIAN)} = 50^{th}$ percentile of of the distribution of the Multi - pathway Acceptable Concentration (MAC)

$\mathrm{ACS}_{\mathrm{ing}\,MEDIAN} = 50^{th}$ percentile of of the distribution of the Acceptable Concentration in the soil medium (ACS) related to the soil ingestion pathway

$\mathrm{ACS}_{\mathrm{dc}\,MEDIAN} = 50^{th}$ percentile of of the distribution of the Acceptable Concentration in the soil medium (ACS) related to the soil dermal conctact pathway

$\mathrm{ACS}_{\mathrm{inhal}\,MEDIAN} = 50^{th}$ percentile of of the distribution of the Acceptable Concentration in the soil medium (ACS) related to the volatile compound and dust inhalation patway

$\mathrm{ACS}_{\mathrm{ing\_w}\,MEDIAN} = 50^{th}$ percentile of of the distribution of the Acceptable Concentration in the soil medium (ACS) related to the water ingestion pathway

$$MACS(5th\_\%ile) = \frac{1}{\frac{1}{ACS_{ing\,5th\_\%ile}} + \frac{1}{ACS_{dc\,5th\_\%ile}} + \frac{1}{ACS_{inhal\,5th\_\%ile}} + \frac{1}{ACS_{ing\_w\,5th\_\%ile}}} \qquad \text{Eq 4.7}$$

where:

$\mathrm{MACS}(5^{th}\ \mathrm{percentile}) = 5^{th}$ percentile of of the distribution of the Multi - pathway Acceptable Concentration (MAC)

$\mathrm{ACS}_{\mathrm{ing}\,5\mathrm{th\_\%ile}} = 5^{th}$ percentile of of the distribution of the Acceptable Concentration in the soil medium (ACS) related to the soil ingestion pathway

$\mathrm{ACS}_{\mathrm{dc}\,5\mathrm{th\_\%ile}} = 5^{th}$ percentile of of the distribution of the Acceptable Concentration in the soil medium (ACS) related to the soil dermal conctact pathway

$\mathrm{ACS}_{\mathrm{inhal}\,5th\_\%ile} = 5^{th}$ percentile of of the distribution of the Acceptable Concentration in the soil medium (ACS) related to the volatile compound and dust inhalation patway

$\mathrm{ACS}_{\mathrm{ing\_w}\,5th\_\%ile} = 5^{th}$ percentile of of the distribution of the Acceptable Concentration in the soil medium (ACS) related to the water ingestion pathway

The human risk is estimated by the use of the risk factor that is defined as the quotient between the estimated concentration in the source medium and the acceptable concentration for that medium. The median and the 95^{th} percentile of the distribution of the risk factor are calculated according to Equation 4.8 and Equation 4.9, respectively.

$$\text{RF(MEDIAN)} = \frac{\text{ESC}}{\text{MACS(MEDIAN)}} \qquad \text{Eq 4.8}$$

$$\text{RF(95th\_\%ile)} = \frac{\text{ESC}}{\text{MACS(5th\_\%ile)}} \qquad \text{Eq 4.9}$$

where:

RF(MEDIAN) = median of the Risk Factor;
RF(95th %ile) = 95^{th} percentile of the distribution of the Risk Factor;
ESC = Estimated Soil Concentration;
MACS(MEDIAN) = 50^{th} percentile of the distribution of the Multi-pathway Acceptable Concentration (MAC) in Soil.
MACS(5th %ile) = 5^{th} percentile of the distribution of the Multi-pathway Acceptable Concentration (MAC) in Soil.

The aggregation into contaminants categories, the risk mapping and vector transformation of both RF(MEDIAN) and RF(95^{th} percentile) are performed with analogy to the deterministic risk assessment described within the previous paragraphs.
An uncertainty indicator is calculated as the difference between the 95^{th} percentile and the median of the Risk Factor, as expressed in Equation 4.10.

$$\text{Uncertainty} = \text{RF(95th\_\%ile)} - \text{RF(MEDIAN)} \qquad \text{Eq 4.10}$$

where:

RF(95th_%ile) = 95^{th} percentile of the Risk Factor distribution;
RF(MEDIAN) = median of the Risk Factor distribution.

With reference to the six contaminant categories, uncertainty indicators are calculated for both RF_{max} and RF_{sum} at each cell of the grid. Analogously to RF, the system can generate raster and vector maps of uncertainty indicators.

The same calculation is performed to generate the Risk Factor for groundwater, where Estimated Groundwater Concentration (EGwC) and Multi-pathway Acceptable Concentration in Groundwater (MACGw) are used in place of ESC and MACS.

CHAPTER 5

Regional risk assessment implemented in SYRIADE

In order to support the risk based inventory of contaminated sites at regional scale, a Spatial Decision Support System (SDSS) was designed and implemented by the EU JRC and Venice Research Consortium. The system is called SYRIADE (Spatial decision support sYstem for RegIonal Assessment of DEgraded land) and allows to rank potentially contaminated sites and mining waste sites at the regional level; to rank risk sources hazard and receptors vulnerability; to integrate risk and socio-economic perspectives for the definition of integrated management areas. SYRIADE is a GIS (Geographical Information System)-based and MCDA (Multi-criteria Decision Analysis)-based SDSS that implements regional and relative risk assessments to support the ranking of potentially contaminated sites where investigation activities are urgently required. In the next paragraphs, the regional risk assessment methodology implemented in SYRIADE is described in detailed, while its application is provided in Chapter 7.

5.1 Regional risk assessment implemented in the SYRIADE DSS

The proposed regional risk assessment methodology can be divided into six different steps which derive from the integration of the steps proposed by Hunsaker and colleagues (Hunsaker et al., 1990) with the relative risk model proposed by Landis and colleagues (Landis, 2005), namely:

- definition of the regional exposure diagram;
- exposure routes risk factor estimation algorithms: definition of the general relative risk model and consequent relative risk equations for the quantification of the relationship between the sources and the receptors throughout the integration of the sources hazard scores, pathway(s) scores and receptor vulnerability scores;
- hazard analysis: qualitative and quantitative description of the source in terms of its hazard by the definition of an appropriate scoring system;

- vulnerability analysis: receptors vulnerability estimation throughout the identification of suitable parameters and the definition of an appropriate scoring system to integrate them;
- pathway relevance analysis: identification of spatial patterns of exposure and assignment of appropriate pathways scores throughout the identification of suitable parameters and the definition of an appropriate scoring system to integrate them;
- regional risk estimation: estimation of the overall relative risk posed by each source through the integration of the risk scores estimated for all impacted receptors in consideration of all considered pathways.

5.1.1 Definition of the regional exposure diagram

The first step for the implementation of the regional risk assessment methodology is the definition of the exposure diagram which identifies the contamination sources, the main release mechanisms, the main potential transport pathways and exposure routes and the receptors which can come in contact with the contamination sources. As reported in Fig 5.1, the chosen potential exposure pathways were the leaching of the contaminants of concern through the vadose zone to the groundwater, the volatilization and wind transport, and the migration of the contamination to the surface water. The main recognized receptors were humans, surface water, groundwater and protected areas. Surface water and groundwater are equally a receptor and a contamination transport medium towards other receptors such as protected areas.

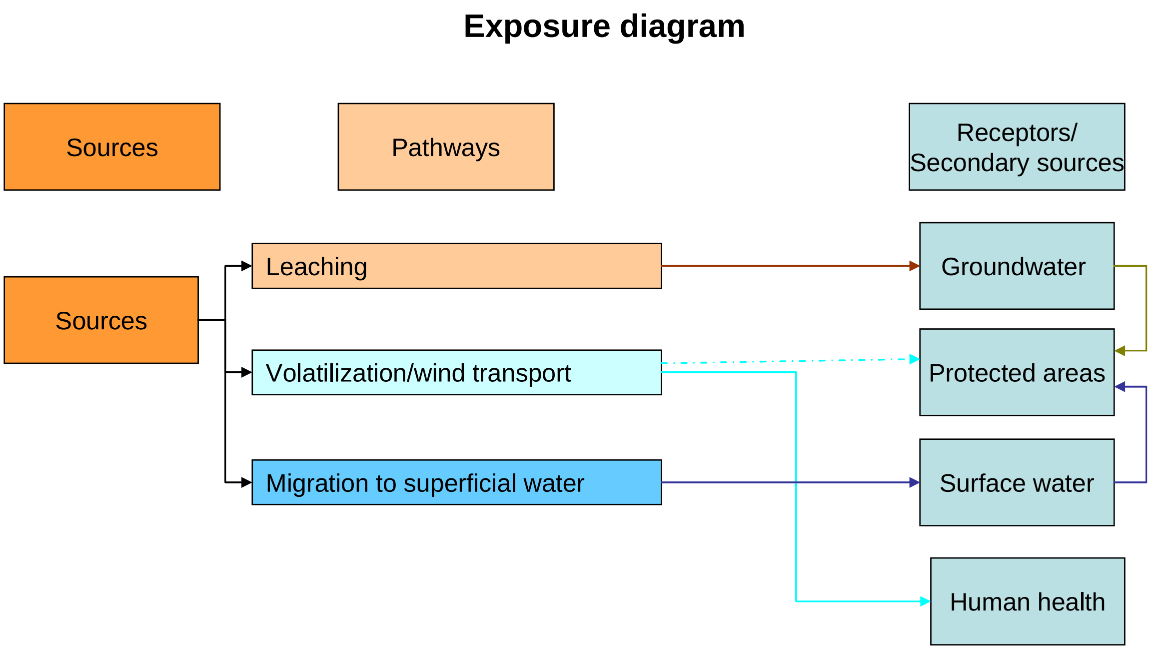

Figure 5.1. Exposure diagram for regional risk assessment, illustrating selected pathways that connect the contamination sources to the relevant receptors.

5.1.2 Definition of the general relative risk model

On the basis of the exposure diagram reported in Figure 5.1, the regional risk assessment components can be identified and the equations used for the estimation of the risk factors posed by the identified contamination sources can be proposed.

The components of the general function for the estimation of the risk factor related to a contamination source S_i, a target T_j and a transport pathway k are reported in Equation 5.1:

$$R_{ikj} = F\left[HS_i, VT_j, f(m_k, n_k), \bar{d}_{ij}\right] \qquad \text{Equation 5.1}$$

where:

R_{ijk} = risk factor related to a contamination source S_i, a target T_j and a transport pathway *k*;

HS_i = hazard score for the source S_i;

VT_j = vulnerability score for the receptor T_j;

$f(m_k)$ = score of the pathway *k* estimated taking into account m parameters characterizing the pathway

$\bar{d}_{ij}$ = vector concerning the distance and/or the direction between the source S_i and the receptor T_j.

Equation 5.1 is general and needs to be adapted to the different identified receptors and the analyzed exposure pathways present in the adapted exposure diagram reported in Figure 5.1.

With reference to the general Equation 5.1, the integration of the risk factor components (i.e. a contamination source S_i, a target T_j and a transport pathway k) is proposed as a product of the considered component scores, as reported in Equation 5.2:

$$R_{ikj} = HS_i * f(m_k, n_k) * VT_j \qquad \text{Equation 5.2}$$

where

HS_i = hazard score for the source S_i, as described in Paragraph 5.1.3. Since more than one activity may have operated in the same site simultaneously, HS_i has an open scale even if its three composing parameters (i.e. toxicity, size and time) can be scored only in a 0-10 scale.

VT_j = vulnerability score for the receptor T_j, which is estimated according to the MCDA procedure described in Paragraph 5.1.4., can vary in a 0-1 scale.

$f(m_k, n_k)$ = score of the pathway k estimated taking into account the m, n parameters characterizing the pathway. The approach for the estimation of the pathway score is obtained adapting the MCDA procedure developed for receptor vulnerability and described in Paragraph 5.1.5. Accordingly, the pathway relevance score can vary in a 0-1 scale.

In the cases where there is a direct contact between the source and the receptor (i.e human health risk factor estimation or groundwater risk factor estimation), Equation 5.2 is performed taking into account only the hazard and the vulnerability scores, since the pathway score is posed equal to 1.

In Table 5.1 the receptors, the exposure pathways, the components of the regional risk assessment engaged in the aggregation functions and the number of the calculation needed for the estimation of the relative risk posed by a contamination source S_i to the potential impacted receptors are reported.

Table 5.1. Identification of the targets, the exposure pathways, the components engaged in the aggregation functions and the number of the calculation needed for the estimation of the relative risk posed by a contamination source S_i to the potential impacted receptors. (S_i= contamination source; GW and gw = groundwater; SW and sw = surface water; HH = Human Health; PA = Protected Area)

Relative risk (R_{ikj})	Target (T_j)	Pathway k	Components which contribute to the aggregation functions	Number of calculations needed for the estimation of the regional risk	Spatial aspects
$R_{i,GW}$	GW		HS_i, VT_{GW}	1 (for each S_i only the hydro-geological homogeneous area it belongs to is considered)	In the region the hydro-geological homogeneous areas are identified (see Paragraph 5.1.4.1)
$R_{i,SW}$	SW		$HS_i,\ VT_{SW}, d\_dist$	1 (for each S_i only the homogenous surface water element it belongs to is considered)	In the region, the homogenous surface water element are identified (see Paragraph 5.1.4.1) Only the homogenous surface water elements within a defined distance, $\bar{d}$, from the sources are considered
$R_{i,v,HH}$	HH	wind	HS_i, VT_{HH}	1 (for each S_i only 1 administrative unit is considered)	In the region the administrative units are identified and characterized by selected parameters
$R_{i,v,PA}$	PA_j	wind	HS_i, VT_{PA}, d_dist	N = number of PAs which can be connected to the source S_i by a distance d	Only the protected areas within a defined distance, $\bar{d}$, from the sources are considered
$R_{i,gw,PA}$	PA_j	gw	$HS_i, VT_{PA},\ f(k_{gw}), d\_dir$	M = number of PAs which are located in the same hydro-geological homogeneous area of the S_i and are downstream with respect to the S_i	In the region some hydro-geological homogeneous areas are identified (see Paragraph 5.1.4.1). Only the direction component of the distance vector $\bar{d}$, is considered (i.e. PAs are downstream with respect to the S_i)
$R_{i,sw,PA}$	PA_j	sw	$HS_i, VT_{PA},\ f(k_{sw}), \bar{d}$	L = number of pairs of S_i and PA which are connected by the river network (within a defined distance d from the river) and when the PA_j is downstream respect to S_i	For each S_i and PA_j only an homogenous surface water element is considered (SW_j). Only the protected areas and the sources within a defined distance, $\bar{d}$, from the homogenous surface water elements are considered

All the regional risk assessment components defined in Table 5.1 need to be characterized by appropriate relevant attributes. These attributes should be identified case by case according to the context in which the regional risk assessment has to be applied and the information available in the specific regional context. The aggregation of the identified attributes for the estimation of the target vulnerability score and pathways score is performed by means of Multi Criteria Decision Analysis (MCDA). MCDA includes a large class of methods for the evaluation and ranking or selection of different alternatives that considers all the aspects of a decision problem involving many actors (Giove et al., 2009). The solution proposed in this thesis is the use of MAVT techniques which are most suitable in cases where alternatives and criteria (attributes in our case) are numerical continuous values to be aggregated in one score, such as in the vulnerability evaluation and pathways score estimation. In Multi-Attribute Utility/Value Theory (MAUT/MAVT), criterion values are first normalized into a common numerical scale by means of a suitable transformation function (or Utility/Value Function). Then criteria are aggregated by a suitable aggregation operator, a function which satisfies a set of rationality axioms.

The aggregation operator selected for the vulnerability evaluation and pathways score estimation is the Choquet integral which is an aggregation function based on a non-additive measure which tries to average the scores given to each different coalition (i.e. subset) of criteria rather than on single criterion scores (Choquet 1953; Murofushi & Sugeno 1987). A detailed description of the MCDA methodology developed for the SYRIADE project is provided in Paragraph 5.1.4.5.

In the following paragraphs the methodologies for the estimation of the regional risk assessment components scores will be presented (i.e. sources hazard score estimation, targets vulnerability scores estimation and pathways relevance scores estimation).

5.1.3 Hazard analysis

According to Van-Camp and colleagues, (2004a), a “potentially contaminated site” is a “site where an activity is or has been operated that may have caused soil contamination”. The aim of the hazard analysis step is the estimation of a hazard score for each potentially contaminated site on the basis of the potentially polluting activities which are or have been operated at the site. Since different industrial activities may be present or were present in the same site now or in the past, in order to estimate the total contaminated site hazard, the composing activities should first be analyzed separately in their hazard, and then the resulting hazards summed together.

The hazard of a potential contamination activity in a site is estimated by evaluation of three main parameters (which can be easily found and evaluated in a screening phase):

- the toxicity of the economic activities present in a site on the basis of the substances potentially produced;
- the size of the industrial activity which can be expressed as production, number of employees, area of the site or volumes of produced waste;
- the time of operation, which defines the length of a considered activity in a site, set to a maximum of 100 years from the present.

In the next paragraphs the three main parameters are discussed in more details and the approach for their final integration, which leads to the estimation of the hazard score associated to each contaminated site located in the region of concern, is presented.

The toxicity of the economic activities present in a site is evaluated in relation to the risk phrases associated to the substances that the activity may produce and release into soil, and the scores of these risk phrases reported in the PRA.MS procedure (EEA, 2005b). The potentially released substances are identified by the DoE industry profiles (UK Environment Agency, 1995) and NORISC studies and reports (NORISC).

The calculation of the toxicity score of an industrial activity is independent from the site-specific parameters and can be determined off-line and assigned from the beginning to a specific industrial activity. The process allows to provide a quantitative score of this parameter in a 0-10 scale.

The substances which can be released into soil are divided in the categories or single substances defined in the NORISC project, such as Total petroleum hydrocarbons, Aliphatic hydrocarbons, Polychlorinated Hydrocarbons, Volatile Halogenated Hydrocarbons, Aromatic Hydrocarbons, Polyciclic aromatic hydrocarbons, Inorganics, Dioxins and furans, Pesticides, MTBE, Chlorinated Aromatic Hydrocarbons, Organotin compounds, Total chlorophenols, Explosives, Phenols, Organolead compounds, Cadmium, Copper, Chromium, Arsenic, Zinc, Nickel, Mercury, Lead. To each substance one or more risk phrases are associated according to the information available in the internet site: http://www.ilo.org/public/english/protection/safework/cis/products/icsc/dtasht/riskphrs/i ndex.htm. Equally, to each risk phrase a score is assigned according to the PRA.MS methodology.

In order to estimate the overall toxicity for a given industrial activity, the following methodology has been developed.

Let:

- n = number of categories;

- N_j = number of substances belonging to the category j (j = 1, …, n);

- ω_{ij} = the PRAMS score for the substance i which is representative of the category j;

- $\omega_{max,j} = \max\limits_{i=1,\ldots,N_j}$ of ω_{ij}, $\forall$ j = 1, …, n;

- $M = \sum_{j=1}^{n} \omega_{max,j}$ = the sum of the maximum scores which can be associated to each category in relation to the substances belonging to that category;

- $x_j \in \{0,1\}$, j = 1, …, n, which means the presence (1) of at least 1 substance within the category or the absence (0) of that category because no substance can be potentially produced by the considered activity.

The substance i, belonging to the category j, characterised by the maximum risk phrase score is chosen as representative substance of the category j and its score is associated to the analysed category, by default. Otherwise, the user can identify the representative substance for the considered category and the related risk phrase score is associated to the whole category.

The score of the overall toxicity for a given industrial activity can be obtained through the application of Equation 5.3:

$$Toxicity = 10 \cdot \frac{\sum_{j=1}^{n} x_j \omega_{ij}}{M} \qquad \text{Equation 5.3}$$

The first term is multiplied by 10 to set the values in a 0-10 scale.

The mechanism can be equally described by a transformation function of the values of the different toxicity scores, for which the maximum is the sum of all the maximum scores which has been assigned to all the categories, and it is scored 10, while the other sums are scored in an interval from 0 to 10. In this approach, the assumption that all the substances potentially produced by an activity are released in the same way, is made. Moreover, the approach is based on the additivity of toxicity of the different substances, which justifies the importance given to the sum of contributions of the toxicity of several substances. Finally, it must be noted that in this linear aggregation of toxicity scores, compensability among the different individual elements is assumed because the scores have the meaning of trade-off ratio or substitution rates (Munda, 2005).

For the size parameter, information on production, number of employees, area of the activity or volume of wastes is taken into account. Ideally the production would provide the most significant information, but when absent, the other parameters are used as its proxy estimation.

Similarly to what was developed in Paragraph 5.1.4.4. for the vulnerability score estimation, a normalisation function is applied to the size parameters values in order to map values from different domain spaces into a single codomain space. The codomain space used for the size score estimation is [0,10], thus all size parameter values are converted into this space by using different normalization functions. The simplest way to define such a function, with respect to expert judgments, is the use of continuous piecewise linear functions identified by the values reported in Table 5.2.

Table 5.2. Size values used in the normalisation function.

Source size in terms of volume of stored wastes	Source size in terms of volume of contaminated soil:	Source size in terms of the area extent of surface soil contamination	Score
m^3	m^3	m^2	
≥ 300000	≥ 1500000	≥ 1500000	10
100001-300000	500001-1500000	500001 - 1500000	8.3
30001-100000	150001-500000	150001 - 500000	6.7
10001-30000	50001-150000	50001 - 150000	5.0
3001-10000	15001-50000	15001 - 50000	3.3
1001-3000	5001-15000	5001 - 15000	1.7
≤ 1000	≤ 5000	≤ 5000	0.3
Source size in terms of Number of employees for service industry	Source size in terms of Number of employees for Production Industry		Score
> 100	> 1000		10
16 - 100	101 - 1000		6.3
≤ 15	≤ 100		2.5

As established in the PRA.MS procedure, the indicators of volume of wastes, employees and area are not used simultaneously, but rather there is a preferential order for their use (first volume, second area, third number of employees).

Finally, time represents the length of operation of each activity on a site. The system considers a maximum timeframe of analysis of 100 years from the present.
To be comparable to the scales of the other two parameters, also the time is easily reported into a 1-10 scale dividing the length of operation of each activity on a site by 10. When site-specific data on operation time are not available, in a conservative approach, the attributed default values is 10.

For each activity, the normalized size value, the toxicity value and the normalised time value are multiplied in order to estimate the activity hazard score.
When more than one activity insists on the same site, the hazard scores of each activity are summed up in order to obtain the total hazard score for the potentially contaminated site as reported in Table 5.3.

By considering the fixed timeframe of 100 years, the maximum score for each activity can be 10x10x10, which defines the worse situation of potential contamination occurring when a single activity with the higher toxicity and a wide dimension is present for all the 100 years. However, since at a site more than one activity can be present, the hazard score have an open scale.
Therefore, by applying this procedure, it is possible to identify the group of sites that have the highest scores of hazard, thus identifying those sites that are surely much more hazardous than the others.

5.1.4 Vulnerability analysis

The term ‘vulnerability of groundwater to contamination’ was introduced by Jean Margat in the late 1960s (Vrba and Zaporozec, 1994). The concept is based on the assumption that the soil-rock groundwater system may provide a degree of protection against contamination of groundwater by ‘self-purification’ or ‘natural attenuation’ (Zaporozec, 2001). This definition recognizes that the vulnerability depends on the characteristics of the site and that differing soil and hydrogeological conditions will give differing levels of vulnerability and afford different degrees of protection. It is also important to note that this concept is independent of the nature of the pollutant (Worrall, 2005).

The DRASTIC method developed by Aller et al. (1985) is one of the most widely used methods to assess intrinsic groundwater vulnerability to contamination (Banton and Villeneuve, 1989; Evans and Mayers, 1990; Navulur, 1996; Rupert, 2001; Al-Adamat et al., 2003; Babiker et al., 2005). The DRASTIC method was developed by the US EPA to be a standardized system for evaluating groundwater vulnerability to pollution (Aller et al., 1985). DRASTIC can be used to set priorities for areas where groundwater monitoring activities can be carried out. DRASTIC is named in consideration of the seven factors considered in the method: Depth to water, net Recharge, Aquifer media, Soil media, Topography, Impact of vadose zone media, and hydraulic Conductivity of the aquifer (Aller et al., 1985).

Although different stand-alone approaches are available for the estimation of groundwater vulnerability to contaminated sites, the same cannot be stated for the other identified regional targets. However, different vulnerability assessment methodologies were developed for human and/or ecological receptors, in order to support the assessment of relative risks posed by contaminated sites at regional scales (EEA, 2004). A review and analysis of the available relative risk assessment procedures for preliminary and simplified risk assessment of (potentially) contaminated sites was published in a report of the European Environment Agency (EEA, 2004) where 27 existing and documented international methodologies were analyzed. The reviewed methodologies are generally applied at the national or regional level for ranking (potentially) contaminated sites on the basis of available data in order to plan priority of actions in terms of detailed site investigation and, in some cases, direct remedial measures. All methodologies have a similar approach for the estimation, in terms of scores, of the sensitivity of the identified regional receptors to the potentially contaminated sites. The review allowed to identify and list the most common parameters used, in the reviewed methodologies, for the estimation of the receptors vulnerability to potentially contaminated sites (EEA, 2005a). This list was the starting point for the identification of the receptor parameters to be used in the vulnerability methodology proposed in this paper. Nevertheless, the spatial evolution of the analyzed receptor vulnerability estimation is one of the main issues tackled in the present work. In fact, along with the emerging recognition of the relevance of spatial aspects in the management of contaminated sites, the need to take into account spatial variability is increasingly being recognized as a further and essential step in sound vulnerability, exposure and risk assessments (Loos et al., 2006, Van-Camp et al., 2004b, Carlon et al., 2008). Indeed, environmental problems are traditionally assessed and presented in non spatial ways although the heterogeneity of the spatial distribution of the variables involved in the vulnerability and risk assessment strongly influence receptor sensitivity analysis, exposure estimations and hence risks (Marinussen and Van der Zee 1996; Hope 2000; Korre et al. 2002; Linkov et al. 2002; Gaines et al. 2005; Makropoulos and Butler 2006; Sundaram, 2008).

The proposed vulnerability assessment methodology for the estimation of receptors sensitivity to contaminated sites at regional scale deals with both spatial entities relationships and entities properties which are integrated by the use of multi criteria decisional methods. The methodology can be divided into 5 different steps, namely:

1. identification of the regional scale receptors;
2. identification of the attributes relevant for the estimation of the receptor vulnerability to contaminated sites;
3. spatial attribution of the identified vulnerability attribute values;
4. normalisation of the vulnerability attribute values;
5. aggregation of the vulnerability attribute values in order to estimate the receptor vulnerability through the use of MCDA methodologies.

In the following paragraphs the whole methodology is presented by explaining in detail each of the five steps presented above.

5.1.4.1 Identification of the regional scale receptors

As reported in Figure 5.1, the selected receptors to contaminated sites at regional scale are humans, surface water, groundwater and protected areas. For each receptor a spatial entity has been defined. As far as humans are concerned, appropriate administrative units (municipality, district, province) can be used, while for surface water and groundwater effective homogeneous surface water elements and hydro-geological homogeneous areas need to be identified, respectively.

In the proposed methodology, the identification of homogeneous surface water elements is made through the use of a suitable catchment scale level within the hierarchical river network as defined at the European level (JRC, 2007): all the segment rivers belonging to the identified catchment scale level compose a single homogeneous surface water element. On the other hand, the hydro-geological homogeneous areas can be defined as aquifers which are not hydraulically linked together (Hubbert, 1940). Finally, protected areas spatial entities can be identified by the expert or decision maker according to the geographical extension and distribution of the protected areas of concern.

5.1.4.2 Identification of the attributes relevant for the estimation of the receptor vulnerability to contaminated sites

For each identified regional receptor, a list of parameters (corresponding to attributes in MCDA) relevant for the estimation of receptor vulnerability to contaminated sites have to be selected, according to the selection criteria and suggestions provided by EEA (2004 and 2005 a and b) and Landis (2005). The parameters selection needs to be adapted to the different case studies taking into account the data available for the region under analysis. The list of parameters selected for the case study area are reported in Table 7.2 and discussed in Chapter 7.

5.1.4.3 Spatial aggregation of the identified vulnerability attribute values

Once the regional targets are identified and the aspects relevant for the estimation of their vulnerability to contaminated sites are selected, it is then necessary to attribute all the selected vulnerability attribute values to the spatial entities representing the identified targets, since not all vulnerability relevant attributes are aspects directly characterizing only the target entity (TE) of concern, but rather are aspects of the surrounding and embedding land.

GIS spatial aggregation techniques are suitable tools to support the collection and assignment of these attributes to the TE (Worboys & Duckham, 2004). For the proposed methodology, a suitable software application within ArcGIS9.2 was developed in order to perform the following spatial aggregations: minimum, maximum, average, sum, widest area, density and area based weighted average.

An example of widest area application is reported in Figure 5.2, where the objective of the aggregation function is to assign to the TE, the groundwater quality class value belonging to the widest intersecting area.

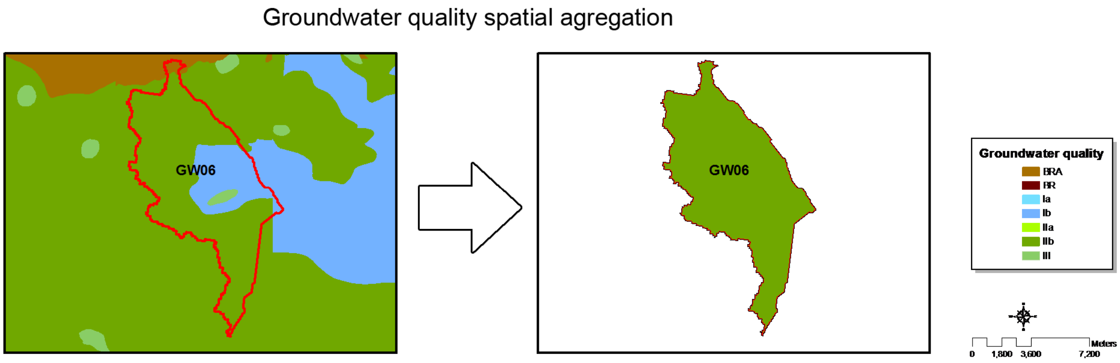

Figure 5.2: Groundwater quality class attribution by means of the widest area aggregation function. The left side shows the layer containing the groundwater quality classes, while on the right site the homogeneous groundwater area (i.e. TE) is identified. Quality classes codes are reported in Table 1.

The attributes values attribution should be performed for each of the relevant parameters identified for the estimation of the regional targets' vulnerability, selecting the most suitable aggregation function between those mentioned above.

5.1.4.4 Normalization of the vulnerability parameters value

The result of the attributes values attribution step is that every TE contains in its attributes set all the attributes values which will be aggregated in order to estimate the vulnerability score for each TE. However, before being able to aggregate the attributes values, a normalization procedure should be performed in order to convert all attributes into the same domain space.

Normalization consists in the application of a defined function in order to map values from different domain spaces into a single codomain space. The codomain space used in the proposed approach is [0,1], thus all TEs' attribute values are converted into this space by using normalization functions shaped according to attributes' relevance in vulnerability evaluation.

The normalization of a finite number of discrete values as well as of a set of labeled classes (e.g. OLD, MID AGE, YUNG) is as simple as the attribution of a score in [0,1] for each domain value. While the normalization of parameters characterized by continuous, possibly infinite, values is concerned with the definition of a scoring function. The simplest way to define such a function, with respect of expert judgments, is the use of continuous piecewise linear functions whose points of segmentation are defined by the experts. Within the proposed methodology, such attribution is guided by normalization tables, as reported in Table 1. The selected classes and their corresponding scores have been set by the experts taking into account PRA.MS (EEA, 2005b) suggestions when possible. In Figure 5.3a and 3b population density and typology normalization functions are derived according to the normalization scores reported in Table 1. In the former case (Fig 5.3a) a monotonic ascendant quasi linear function is used because Human Health’s vulnerability increases with population density; in the latter case (Fig 5.3b) the function, besides also being monotonic, has a descendent tendency, this because Surface Water’s vulnerability decreases as Surface Water’s flow rate increases.

Table 5.2: Normalization function class values

Human health				
Parameter	**Spatial aggregation function**	**Class**	**Description**	**Score**
Population density (P.D.)	-	>= 2000 people/km2	High density	1
		1000 people/km2	Medium density	0,6
		250 people/km2	Low density	0,2
Land use (L.U.)	-	Residential	Urban fabric	1
		Agricultural/Livestock	Agricultural areas	0,8
		Park/School	Green urban areas and sport and leisure facilities	0,6
		Forest	Forest	0,4
		Industrial/Commercial	Industrial or commercial units	0,2
		Isolated area	Isolated area	0
Percentage of vulnerable groups (%V.G.)	-	> 40	High	1
		32-40	Medium-high	0,7
		22-31	Medium	0,5
		<22	Low	0,3

Ground water				
Parameter	**Spatial aggregation function**	**Class**	**Description**	**Score**
Degree of isolation from soil surface (Iso.)	widest area	a – low or lacking	degree of isolation less than 15 m thick till or 5 thick clay in a overburden	1
		b – mid	degree of isolation between 15 m to 50 m thick till or between 5 to 10 m thick clay in a overburden	0,5
		c – high	degree of isolation more than 50 m thick till or more than 10 m thick clay in a overburden	0
Number of water bearing layers (B.L.)	widest area	1	one layer	0
		2	two layers	0,25
		3	three layers	0,5
		4	four layers	0,75
		> 5	more than five layers	1
Quality class (Qua.)	widest area	BRA	the main groundwater horizon do not exist/lack of information	1
		BR	the main groundwater horizon do not exist	0
		I a	water quality good and stable, do not require conditioning	1
		I b	water quality good, but can be unstable because of insufficient isolation layer, water do not require conditioning	0,75
		II a	water quality moderate, water require simple sanitation, (according to procedures used in standard water conditioning station in water supply systems: removal of iron, manganese, aeration), the quality is stable	0,5
		II b	water quality moderate, water require simple sanitation (according to procedures used in standard water conditioning stations in water supply systems: removal of iron, manganese, aeration), the quality is potentially unstable because of insufficient isolation layer	0,25
		III	water quality bad, water require complicated sanitation (according to procedures beyond those used in the standard station of water conditioning in the water supply system)	0
Use class (Use)	more conservative value (maximum normalised value)	Municipal use		1
		Private well	use not specified	0,9
		Irrigation	food, vegetables	0,85
		Other use not specified	Golf, parks, no food, vegetables, etc.	0,55
		Industrial use		0,3
		Non used		0,1
		No groundwater body present		0

Protected areas				
Parameter	**Spatial aggregation function**	**Class**	**Description**	**Score**
Protection typology (Typ.)	-	State	Protected areas at state level	1
		Region	Protected areas at regional level	0,8
		Natura 2000	Sites identified by Natura 2000 database not included in the first two classes	0,6
		Ecological areas	Ecological area protected at local level	0,5
		Biogenic monuments	Biogenic natural monuments protected at local level	0,4
		Abiogenic monuments	Abiogenic natural monuments protected at local level	0,3
		Cultural heritage	Cultural heritage monuments protected at local level	0,2
Extension (Ext.)	-	>= 2000000 m2	Huge extension	1
		1500000 m2	Large extension	0,83
		500000 m2	Medium extension	0,67
		50000 m2	Small extension	0,3
		5000 m2	Tiny extension	0,03

Surface water				
Parameter	**Spatial aggregation function**	**Class**	**Description**	**Score**
Quality class (Qua.)	widest area	Water I class (I)	point on the stream, water body or its part, where was determined that the water meets drinking water standards, food industry needs or other industries requiring drinking water quality and the quality for living conditions of Salmonidae species	1
		Water II class (II)	point on the stream, water body or its part where it was determined that the water is usable for living conditions for farming fish other than Salmonidae species, husbandry use, bathing activities and recreational activities.	0,6
		Water III class (III)	point on the stream, water body or its part, where it was determined, that the water is usable for industry purposes except for the industries requiring drinking water quality, watering purposes in agriculture and horticulture purposes.	0,3
		Water non class (N)	point on the stream, water body or its part, where was determined that the water is not usable in any of the use categories because of its above limits contamination.	0
Typology (Typ.) surface water flow rate (m3/s)	more conservative value (maximum normalised value)	0	Small to moderate stream	1
		10	Moderate to large stream	0,8
		30	Large stream or river	0,6
		100	Large river, coastal tidal waters, shallow ocean zone, great lake	0,5
		1000	Very large river, moderate ocean zone	0,3
		>= 2000	Extreamely large river, ocean zone	0
Use class (Use)	more conservative value (maximum normalised value)	Potable		1
		Recreational (Swimming/Bathing)		0,8
		Piscicolture (salmonid/Shellfish)		0,75
		Irrigation (Food crop)		0,65
		Piscicolture (Cyprinid)		0,6
		Irrigation (No food crop)		0,55
		Industrial/Other		0,3
		Non used		0,2

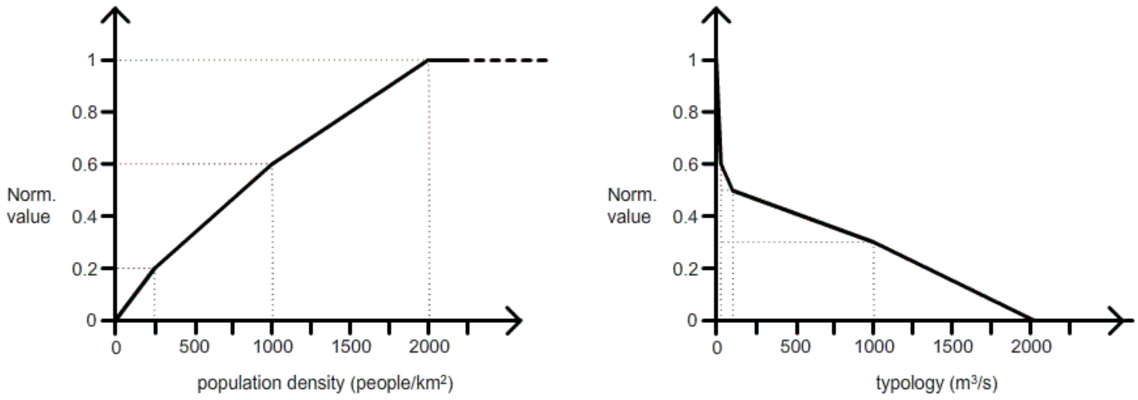

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Figure 5.3: Normalization function examples applied in the case study.

5.1.4.5 Aggregation of the vulnerability parameters values into the receptor vulnerability score, through the use of MCDA methodologies

After applying the normalization functions, vulnerability estimation can be performed as follows. Given a TE $\boldsymbol{Z}$ with attributes set $\boldsymbol{A}$ composed by $\boldsymbol{n}$ attributes, $\boldsymbol{A} = \{\boldsymbol{a_1}, \dots, \boldsymbol{a_n}\}$, vulnerability is evaluated by applying a score function $\boldsymbol{s}: \boldsymbol{A} \rightarrow [\boldsymbol{0}, \boldsymbol{1}]$ to the different alternatives. Vulnerability to contaminated sites is not a linear function, therefore experts knowledge about synergic and redundant interactions between attributes need to be embedded in the function to made it consistent with their thoughts. MCDA methods based on non additive measures are able to fulfill these requirements. As a drawback, they require many more parameters than other methods but they can be used to approach many cumbersome problems.

Accordingly, a measure defined on a set $\boldsymbol{S}$ is a set function assigning a value to each element of the power set of $\boldsymbol{S}$. Measures can be used in aggregation functions to assign a weight to each possible coalition of criteria. A coalition of criteria represents a subset of criteria being “high” at the same time: e.g., in vulnerability assessment criteria are considered “high” when they strongly increase the overall vulnerability score. The importance of a coalition can be greater, equal, or less than the sum of the importance (weights) of each criterion included in the coalition. This way non additive measures are able to deal with synergic and adversarial interactions among criteria. Within the proposed methodology, for the purposes of vulnerability to contaminated sites evaluation, non additive non monotonic measures were applied. A measure *m* on the set N, $m: \mathcal{P}(N) \rightarrow [0,1]$, is defined as non additive monotonic when $\forall\, S, T \subseteq N$ and the following conditions are met:

- $m(\emptyset) = 0$
- $\forall\, S, T \subseteq N: S \subseteq T \Rightarrow m(S) \leq m(T)$
- $m(N) = 1$

The first and the third conditions are intuitive border conditions, while the second one is a monotonicity constraint, that intuitively states that when more (benefit) criteria are satisfied, the global satisfaction cannot decrease.

A non additive measure is classified as:

- additive if: $m(S \cup T) = m(S) + m(T),\ S \cap T = \emptyset$
- sub-additive if: $m(S \cup T) < m(S) + m(T),\ S \cap T = \emptyset$
- super-additive if: $m(S \cup T) > m(S) + m(T),\ S \cap T = \emptyset$

A sub-additive measure models a redundant effect, while a super-additive models a synergic effect and an additive measure degenerates into the WA case.

As stated before, the proposed approach concerns the use of non monotonic measures, this means that only the $m(\emptyset) = 0$ constraint still holds. It still holds, since the null option (no cost and no benefit) is the border between acceptable (with measure greater than zero) and not acceptable alternatives (with negative measures), so it is characterized by an indifferent judgement.

Since not limited to monotonic measures, the method differs from the common approach, and also very few literature exists in the field of non monotonic measures (De Waegenaere & Wakker, 2001; Murofushi & Sugeno, 1994; Cardin & Giove, 2008]. By removing the monotonicity constraint, in the line of principle, a decision maker can model a criterion that is neither a benefit nor a cost, but whose nature (benefit or cost) depends on the coalition to which it belongs. It would be then possible to give two different interpretations to monotonicity violations:

- there is a partial irrationality or uncertainty of the user/decision maker;

- there are conflicting or synergic effects among the attributes.

Within the proposed methodology, measures must be defined by environmental experts. This definition consists in the choice of a score related to each possible coalition of criteria. The most simple and used method to perform such process and elicit experts' thoughts is the use of ad-hoc questionnaires to be filled in. In the presented approach such questionnaires have been used in the shape of simple tables to fulfill: indeed, tables are the basic type of questionnaires which can be used in measures definition. The proposed questionnaires are reported in Table 7.3 in the case study application section: each row of the questionnaire's tables represents a different criteria' situation, where a value of 1 should be interpreted by the expert as the situation when the criteria values are in their highest class (i.e. maximum influence in the vulnerability estimation) and a value of 0 as the situation when criteria are in their lowest class (i.e. minimum influence in the vulnerability estimation). Score values are integer numbers in the [0,100] closed set, given by the expert to the presented different combinations of highest presence or absence of the considered criteria.

Once the measures to be used are defined, an aggregation function parameterized on this measure should be used. The developed MCDA aggregation function mimics the behavior of WA in linear environments by using the Choquet integral which can be seen as a non linear version of WA. It is a very general operator that can also mimic other aggregation operators such as minimum, maximum, OWA, etc.

The Choquet integral is defined as follows (Choquet, 1953):

Let μ be a measure on X, whose elements are denoted $x_1, \dots, x_2$ here. The discrete Choquet integral of a function $f: X \to \mathbb{R}^+$ with respect to μ is defined by

$$C_\mu(f) := \sum_{i=1}^{n} \left(f\left(x_{(i)}\right) - f\left(x_{(i-1)}\right)\right) \mu\left(A_{(i)}\right)$$

Where $\cdot_{(i)}$ indicates that the indices have been permuted so that $0 \leq f(x_{(1)}) \leq \cdots \leq f(x_{(n)})$, $A_{(i)} := \{x_{(i)}, \dots, x_{(n)}\}$, and $f\left(x_{(0)}\right) = 0$.

In the presented case the f function is set to be the Identity function such that $f(x) = x$, therefore the results of the Choquet integral are directly related only to the measure μ and the values in X.

The idea underlying the Choquet integral is that different coalitions can be present, each with a different amount, in the same scenario and that the scenario's score is obtained by weighting the coalitions' related scores by their amount (see Murofushi & Sugeno, 1989, for a complete example).

5.1.5 Pathway relevance analysis and sources-targets relationship assessment

According to Figure 5.1 and Table 5.1, the identified regional receptors can be reached by the contamination through a direct contact with the contaminated sites and/or they can be impacted by contaminated surface water and groundwater. In the first case no further pathway analysis needs to be performed since the only requirement, which needs to be verified is the spatial overlapping between sources and receptors. In the second case, the pathway analysis can be split in two parts, the pathway relevance analysis and the sources-targets relationships assessment.

The methodological approach for the estimation of the pathway relevance score is similar to the one developed for the receptor vulnerability estimation reported in Paragraph 5.1.4. The pathway relevance estimation can be defined as the analysis of the intrinsic characteristics of the pathway which control the transport of a chemical of concern from the potentially contaminated sites to an exposed target. The analysis entails the following steps:

- identification of the regional pathways;
- identification of the parameters (called attributes in MCDA) relevant for the estimation of the pathway relevance;
- spatial attribution of the identified pathway parameter values;
- normalisation of the pathway parameter values;
- aggregation of the pathway parameter values in order to estimate the pathway score through the use of MCDA methodologies.

According to Figure 1, in the proposed Regional Risk Assessment, the identified regional pathways are of three types:

- wind;
- surface water;
- groundwater.

Similarly to what is defined in Paragraph 5.1.4. for the vulnerability score estimation, for each identified pathway a spatial entity needs to be defined. In the case of the wind pathway, the potential impacted receptors are those receptors located within a buffer of radius d around each source. A default value of the radius can be assigned, but the user of the DSS can change the predefined value. Thus the spatial entity representing the wind pathway is the buffer area around the potentially contaminated sites.

As far as groundwater and surface water are concerned, the homogeneous surface water elements and hydro-geological homogeneous areas identified in the vulnerability assessment can be used as spatial entities for the definition of the surface water and groundwater pathway scores. As described in Paragraph 5.1.4., the identification of homogeneous surface water elements is made through the use of a suitable catchment scale level within the hierarchical river network as defined at the European level (JRC, 2007): all the river segments belonging to the identified catchment scale level compose a single homogeneous surface water element. On the other hand, the hydro-geological homogeneous areas can be defined as aquifers which are not hydraulically linked together (Hubbert, 1940). Hydrogeological experts should identify the hydrogeological homogeneous areas which are present in the region under analysis and develop the relative GIS map.

For each identified regional pathway, a list of parameters (called attributes in MCDA) relevant for the estimation of the importance of the pathway to contaminated sites were selected according to EEA (2004 and 2005) and Landis (2005). The selected parameters need to be adapted to the different case study taking into account the data available for the region under analysis. The methodologicaly of the spatial attribution, normalisation and MDCA aggregation steps are explained in Paragraph 5.1.4.5.

Once the pathway relevance estimation is completed, the following assessment concerns the identification of the receptors reached by the contamination through the three identified regional pathways. In the wind pathway assessment, the targets reached by the contaminated sites are those included within the buffer of radius d around each source.

The identification of the receptors reached by the contamination through the surface water pathway is a critical aspect that requires specific spatial management. As explained in Table 5.1, every source can have an impact to near protected areas, only if both of them are included in a buffer area around the river and if the protected area is downstream with respect to the source. The system spatially assesses this relation, thanks to GIS functionalities. First of all, a buffer area is defined for every river in the region. For each source and protected area only the river related to the catchment where the source or the protected area is located is taken into account, among all the rivers buffers where the source or the protected area can be included.

Then, in order to identify all the source-receptor pairs (i.e. all the downstream protected areas impacted by each source), a surface water flow network has to be built. To this end, the set of polylines representing the river segments oriented according to the flow direction (identified during a vectorialisation process) have to be appropriately linked by a tool, called Rivers Graph, specifically developed during the project and implemented in ArcGIS.

According to the “Rivers Graph” tool capabilities, each source-downstream receptor pair is identified and saved in a table used to associate to each identified source-receptor pathway the appropriate value of the surface water pathway relevance. In fact, since the path which links the source with the receptors along the river can include more than one homogeneous surface water element, a suitable surface water pathway relevance score should be assigned to the whole path. The selection of the surface water pathway relevance score can be made according to one of the following approaches:

- the pathway relevance value belonging to the longest river segment is taken as representative value of the whole river pathway;
- the more conservative pathway relevance value is taken as representative of the whole river pathway, in a cautionary approach;
- an averaged value of the pathway relevance values, weighted by the length of the related river segments within the whole river pathway, is considered.

Finally, according to Table 1, the identification of the receptors reached by the contamination through the groundwater pathway is performed selecting the receptors which are located in the same hydro-geological homogeneous area of the generic contaminated site S_i and are downstream with respect to S_i. In order to estimate the groundwater flow direction, the map of the spatial distribution of the groundwater table levels needs to be developed.

5.1.6 Regional risk estimation

Once the risk factor R_{ikj}, posed by each source S_i for each pathway and potentially impacted receptor is estimated according to the procedure described in Paragraph 5.1.2, the regional relative risk posed by the source S_i is calculated through the integration of the risk factors estimated for all the impacted receptors (i.e. all the risk factors reported in Table 5.1 for an analyzed source S_i with respect to the active pathways). To this purpose, for each contamination source, the sum of the risk factors calculated for each affected receptor is performed, to obtain the value of the regional risk of each source (RS_i), as in Equation 5.4:

$$RS_i = \sum_{jk} R_{ikj} \qquad \text{Equation 5.4}$$

where:

R_{ikj} = the risk factor related to a contamination source S_i, a target T_j and a pathway k, as calculated in Equation 5.2.

The user, who can visualize separately the different exposure pathways that contribute to the regional risk of each source, can compare these contributions, in order to verify which one affects more the regional risk score.

Equally, at the end of the assessment, the calculation of the regional risk score for each source allows the final straightforward ranking of potentially contaminated sites in the region.

Finally, the information concerning risk factors and regional risk scores are provided to the user in separate information tables and GIS maps, in order to facilitate their interpretation and comparison. In order to support the national and regional authorities in the evaluation of the administrative unites more affected by potentially contaminate sites, also a NUTS regional relative risk calculation is proposed as the sum of all regional risk scores estimated for all the potentially contaminated sites located in each different NUTS.

CHAPTER 6

DESYRE application to Porto Marghera case study

The application of the spatial risk assessment methodology developed in the DESYRE project was described in Carlon et al, 2008. The next paragraphs report the case-study section of this paper.

6.1 General conceptual model of the site and input data description

The DESYRE software was tested on a 450 hectares subarea of a contaminated megasite of national interest, located in Porto Marghera and bordering the Venice lagoon (Italy). The megasite's total land surface covers approx 2000 hectares and is occupied by industrial enterprises, commercial port areas, communication roads, railways, services and lagoon canals. According to the national regulation, and with the objective of developing a Master Plan for the remediation of the whole area, a preliminary characterisation of the megasite was performed based on a regular sampling grid of 100 meters. The soil showed a complex and spread contamination, mainly represented by PAHs, amines, halogenated organic compounds including dioxins, and metals (such as As, Cd, Pb, Zn) (Municipality of Venice, 2003).

Due to the wide extension of the impacted area and the presence of several classes of pollutants, in the short-medium time frame (next 25 years) the overall remediation of the site would be not feasible, and spatial priorities for remediation had to be defined at the site based on the potential risk for human health.

To cope with this objective, the DESYRE software was applied and the results were fed into the definition of a preliminary Master Plan for the remediation of the site. The results are described in the following paragraphs and concern the conceptual model formulation, selection of contaminants of concern, exposure and toxicological values, risk characterisation and allocation of remediation technologies, respectively.

The general conceptual model of the Porto Marghera site consisted of three models representing the hydrogeology, contamination and exposure, respectively (Municipality of Venice, 2003).
As far as the hydrogeology model, the site is placed over a coastal multiple aquifer system in contact with the lagoon surface water. A waste filling about 4 meters thick, including red bauxitic mud and/or black organic sludge, is located on a first impermeable layer consisting of Holocene deposits of lagoon mud flats (so called barene) and consolidated silt clay (called caranto). In between this first impermeable layer and a bottom clayey Pleistocene sediment (i.e. a second impermeable layer), a semi-confined aquifer exists (Figure 2 in Critto et al., 2006).

As far as the contamination is concerned, in the top soil (i.e. the filling material layer) several classes of pollutants were found: amines, chlorobenzenes, chloronitrobenzenes, chlorophenols, dioxins (PCDDs/Fs), aliphatic hydrocarbons, polynuclear aromatic hydrocarbons (PAHs), metals, metalloids and inorganic anions (Municipality of Venice, 2003). Metals and metalloids showed the highest concentration levels and the widest spread of contamination. The analyzed soil samples displayed high concentrations of arsenic (hot spots of 900 mg/Kg d.w.), chromium (hot spots of 4200 mg/Kg d.w.), cadmium (hot spots of 900 mg/Kg d.w.), copper (hot spots of 3000 mg/Kg d.w.), mercury (hot spots of 130 mg/Kg d.w) and lead (hots spots of 26000 mg/Kg), where hot spot is defined as a local, restricted area where the concentration of one (or more) contaminant(s) is especially high, i.e. under common exposure conditions it is likely to lead to a relevant risk to human health or other receptors (Carlon et al., 2004). The analyses of the semi-confined aquifer showed a widespread contamination similar to soil, mainly of more soluble contaminants, likely due to soil leaching processes. For reasons of simplicity, this demonstration case study will only address soil contaminants. Notwithstanding, leaching of soil contaminants to groundwater and groundwater human consumption will be considered as potential pathway of human exposure to soil contaminants.

As far the exposure conceptual model, the diagram in Figure 6.1 represents the exposure pathways considered in the risk assessment. It can be noted that humans on site and the Venice Lagoon were the considered receptors. Off-site receptors exposed by consumption of contaminated groundwater were not considered, because the groundwater flow is directed towards the lagoon water body. For the sake of simplicity, the exposure of the lagoon water through contact with contaminated groundwater was not included in the presented case study. The complex relation between the two water bodies, which is affected by sea tidal effects, has been already investigated by the authors (Critto et al., 2004) and could not been simulated by the modelling provided by the DESYRE system. Accordingly, its description is beyond the scope of this thesis.

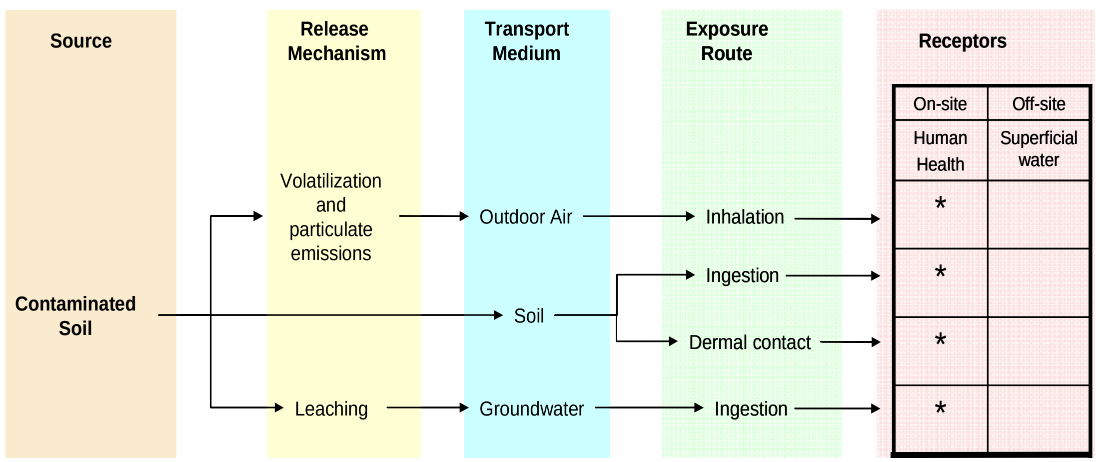


Figure 6.1. The exposure conceptual model for the Porto Marghera case study.

6.1.1 Selection of the substances of concern

Based on the risk scoring system described in the methods section, 18 Chemicals of Concern (CoC) were selected out of 325 analysed chemicals. They include inorganic substances and Non Halogenated Semi-Volatile Organic Carbons (NH-SVOCs) (Table 6.1). It is notable that halogenated semi volatile contaminants (H-SVOCs) and volatile contaminants (VOCs) in general did not show significant risk contributions in soil. However, it should be borne in mind that this demonstration case study is limited to the soil impacted media, whereas the assessment of the groundwater contamination would lead into consideration a series of more soluble substances, in particular chlorinated hydrocarbons, that are not present in soil at significant concentration.

Table 6.1.The contaminants of concerns (CoC) selected according to the Concentration-Toxicity screen method (USEPA, 1989) and divided in the following two categories: Inorganics and Nonhalogenated Semivolatile Organic Compounds (NH-SVOC), (FRTR, 2002). For each substance, the % ratio between the hazard score (HS) and the total hazard score ($HS_{\_tot}$) for carcinogenic and non carcinogenic properties have been reported both considering the case of maximum concentration and mean concentration in soil.

	Max Concentration		Mean Concentration		Category
Subsatances	$\%HS_{carc}/HS_{tot}$	$\%HS_{no\_carc}/HS_{tot}$	$\%HS_{carc}/HS_{tot}$	$\%HS_{no\_carc}/HS_{tot}$	
Arsenic	17.7		36.3	3.2	Inorganics
Cadmium	2.2	41.1	1.5	5.2	Inorganics
Chromium	63.0		60.3		Inorganics
Manganese		44.1		83.6	Inorganics
Mercury				5.0	Inorganics
Lead		0.9			Inorganics
Copper		1.8			Inorganics
Vanadium		2.5		2.1	Inorganics
Zinc		8.9			Inorganics
Benzo(a)anthracene	1.3				NH-SVOC
Benzo(a)pyrene	13.5		1.0		NH-SVOC
Benzo(b)fluorantene	0.9				NH-SVOC
Dibenzo(a,h)anthracene	0.6				NH-SVOC
%HS_cumulative	99.2	99.3	99.1	99.1	

6.1.2 Exposure and toxicological values

Selected CoC were mapped by means of geostatistical interpolation methods on the basis of a 25 meters grid (estimation at each node of the grid, which gives a raster map with a 25 m^2 resolution). The contaminants spatial distribution maps are not reported in this paper.

Due to the well established vocation of the Porto Marghera area to host infrastructures and industrial facilities, the exposure assessment was performed for the industrial use scenario. The corresponding exposure parameters are reported in Table 6.2.

Table 6.2. Exposure parameters: from left to right, the columns report the parameter acronym, the parameter definition, the unity of measure, the parameter representative value used in the deterministic exposure modelling, the type of distribution considered in the probabilistic approach, the parameters statistics describing the distribution.

Parameter	Description	Unity of measure	Representative value	Distribution	Minimum	Maximum	Media	Standard Deviation
EF	Exposure Frequency	(days/year)	2.50E+02	Uniform	2.00E+02	3.00E+02		
ED	Exposure Duration	(year)	2.50E+01	Uniform	2.40E+01	3.60E+01		
BW	Body Weight	kg	7.00E+01	Log-Normal			4.26E+00	1.95E-01
CR_{ing-s}	Soil ingestion rate	(mg/day)	5.00E+01	Uniform	3.00E+01	1.00E+02		
AT_c	Defined averaging time for carcinogenic substances	(years)	7.00E+01	Uniform	7.00E+01	8.00E+01		
AT_{nc}	Defined averaging time for non-carcinogenic substances (= ED)	(years)	2.50E+01	Uniform	2.40E+01	3.60E+01		
SA	Exposed Skin Surface area	cm2	3.16E+03	Log-Normal			8.63E+00	1.81E-01
AF	Soil to skin adherence factor	mg/cm2	1.00E+00	Uniform	1.00E+00	2.80E+00		
CR_{inhal}	Inhalation rate for volatile substances and particulate emissions from soil	(m3/day)	2.00E+01	Log-Normal			1.77E+00	5.20E-01
CR_{ing-w}	Water ingestion rate	(l/day)	1.00E+00	Log-Normal			1.87E-01	5.00E-01
TR_c	Acceptable risk for carcinogenic substances	-	1.00E-06	Constant				
TR_{nc}	Acceptable risk for non-carcinogenic substances	-	1.00E+00	Constant				

The toxicological assessment was carried out on the basis of the most conservative toxicological values from the IRIS (USEPA, 2002) and RAIS (ORNL, 2002) databases.

For the sake of demonstration of input parameters, the chemical-physical and toxicological parameters of one of the selected CoC (Benzo(a)pyrene) are reported in Table 6.3 and Table 6.4, respectively. Similarly to Table 6.2, Table 6.3 and 6.4 report both the values used in the deterministic approach and those applied in the probabilistic approach.

Table 6.3. Chemical-physical parameters for Benzo(a)pyrene: the values used for the deterministic approach are reported under the column “Representative values”, while the probability distributions considered for the probabilistic approach described by their statistics are reported under the columns “Distribution”, “Minimum”, “Maximum”, “Mean”, “Standard Deviation” and “Mode”.

Substance	Parameter	Description	Unity of measure	Representative values	Distribution	Minimum	Maximum	Media	Standard Deviation	Mode
Benzo(a)pyrene	H	Henry’s law constant	(cm3-H2O/cm3 - air)	4.60E-05	Uniform	3.70E-05	5.56E-05			
	k_d soil	Soil-water sorption coefficient	(l/kg)	1.02E+04	Log-Normal			1.16E+01	1.01E+00	
	D^{wat}	Diffusion coefficient in water	(cm2/s)	9.00E-06	Triangular	7.20E-06	1.08E-05			9.00E-06
	D^{air}	Diffusion coefficient in air	(cm2/s)	4.30E-02	Triangular	3.44E-02	5.16E-02			4.30E-02
	DA	Relative adsorption factor for soil dermal contact	-	1.00E-02	Triangular	1.10E-01	1.50E-01		3.00E+00	1.30E-01
	ABS_{ing-s}	Gastro Intestinal absorption factor	-	1.00E+00	Constant					

Table 6.4. Toxicological parameters for Benzo(a)pyrene: the values used for the deterministic approach are reported under the column "Representative values", while the probability distributions considered for the probabilistic approach described by their statistics are reported under the columns "Distribution", "Minimum", "Maximum", "Mean", "Standard Deviation" and "Mode".

Substance	Parameter	Description	Unity of measure	Representative values	Distribution	Minimum	Maximum
Benzo(a)pyrene	RfD_o	Oral reference dose	[mg/(kg·day)]				
	RfD_{inhal}	Inhalation reference dose					
	RfD_d	Dermal reference dose					
	SF_o	Oral slope factor	$[mg/(kg \cdot day)]^{-1}$	7.3	Uniform	0.73	73
	SF_{inhal}	Inhalation slope factor		3.1		0.31	31
	SF_d	Dermal slope factor		23.5		2.35	235

6.2 Risk characterisation

The human health risk assessment has been estimated by the use of both the deterministic and the probabilistic methods.

6.2.1 Deterministic risk assessment

The Multi-pathway Acceptable Concentrations in Soil, MACS, were calculated, for all the considered substances, according to Equation 4.3 and reported in Table 6.5 with the acceptable concentrations in soil calculated for specific exposure routes (ingestion of soil, inhalation of volatile substances and particulate, dermal contact with soil and ingestion of water contaminated by leaching of contaminants from soil).

Table 6.5. Estimated acceptable concentrations in soil for the considered exposure routes (ingestion of soil (ACS_IS), dermal contact with soil (ACS_DCS), inhalation of volatile substances and particulate (ACS_IVS) and ingestion of contaminated groundwater by leaching of contaminants from soil (ACS_IGW)) and the Multi-pathway Acceptable concentration in Soil (MACS). The acronyms C, NC and CNC are referred to Carcinogenic, Non Carcinogenic and both Carcinogenic - Non Carcinogenic contaminant toxicological effects respectively.

CATEGORY	CONTAMINANT	TOXICITY	ACS_IS	ACS_DCS	ACS_IVS	ACS_IGW	MACS
			mg/kg_{ss}	mg/kg_{ss}	mg/kg_{ss}	mg/kg_{ss}	mg/kg_{ss}
NH_SVOCs	Benzo(a)anthracene	C	7,84E+00	3,85E+00	5,64E+01	3,80E+01	2,32E+00
	Benzo(a)pyrene	C	7,84E-01	3,85E-01	6,26E+00	9,73E+00	2,42E-01
	Benzo(b)fluorantene	C	7,84E+00	3,85E+00	6,30E+01	1,17E+02	2,43E+00
	Dibenzo(a,h)anthracene	C	7,84E-01	3,85E-01	6,37E+00	3,62E+01	2,47E-01
Inorganics	Arsenic	CNC	3,82E+00	2,47E+01	3,98E-01	9,29E-01	2,57E-01
	Cadmium	CNC	1,02E+03	1,62E+02	3,26E+00	9,34E+01	3,08E+00
	Chromium	CNC	7,84E+02	3,10E+02	4,74E-01	1,83E+01	4,61E-01
	Manganese	NC	9,40E+04	5,95E+04	1,02E+02	7,45E+03	1,00E+02
	Mercury	NC	6,13E+02	6,79E+02	1,50E+02	3,95E+01	2,85E+01
	Lead	NC	7,15E+03	1,13E+05	2,49E+04	7,83E+03	3,16E+03
	Copper	NC	1,02E+06	1,62E+07	3,55E+06	5,32E+05	3,12E+05
	Vanadium	NC	1,43E+04	2,26E+03	4,97E+04	1,74E+04	1,70E+03
	Zinc	NC	7,15E+03	1,94E+06	2,13E+06	4,64E+04	4,13E+04

The ratio between the acceptable concentration and observed concentration (Equation 4.5) was calculated at each cell of the CoCs concentration maps, which resulted in risk factor maps for every selected CoC. Based on the selection at each cell of the maximum value (RFmax) or the sum of values (RFsum) of Risk Factors, RFmax and RFsum maps for each contaminant categories were generated, respectively.

Based on the classification of the Risk Factor into five classes, raster maps were transformed into vector maps.

The resulting Risk Factors vector maps for NH_SVOCs and inorganics are reported in Figure 6.1 and 6.2.

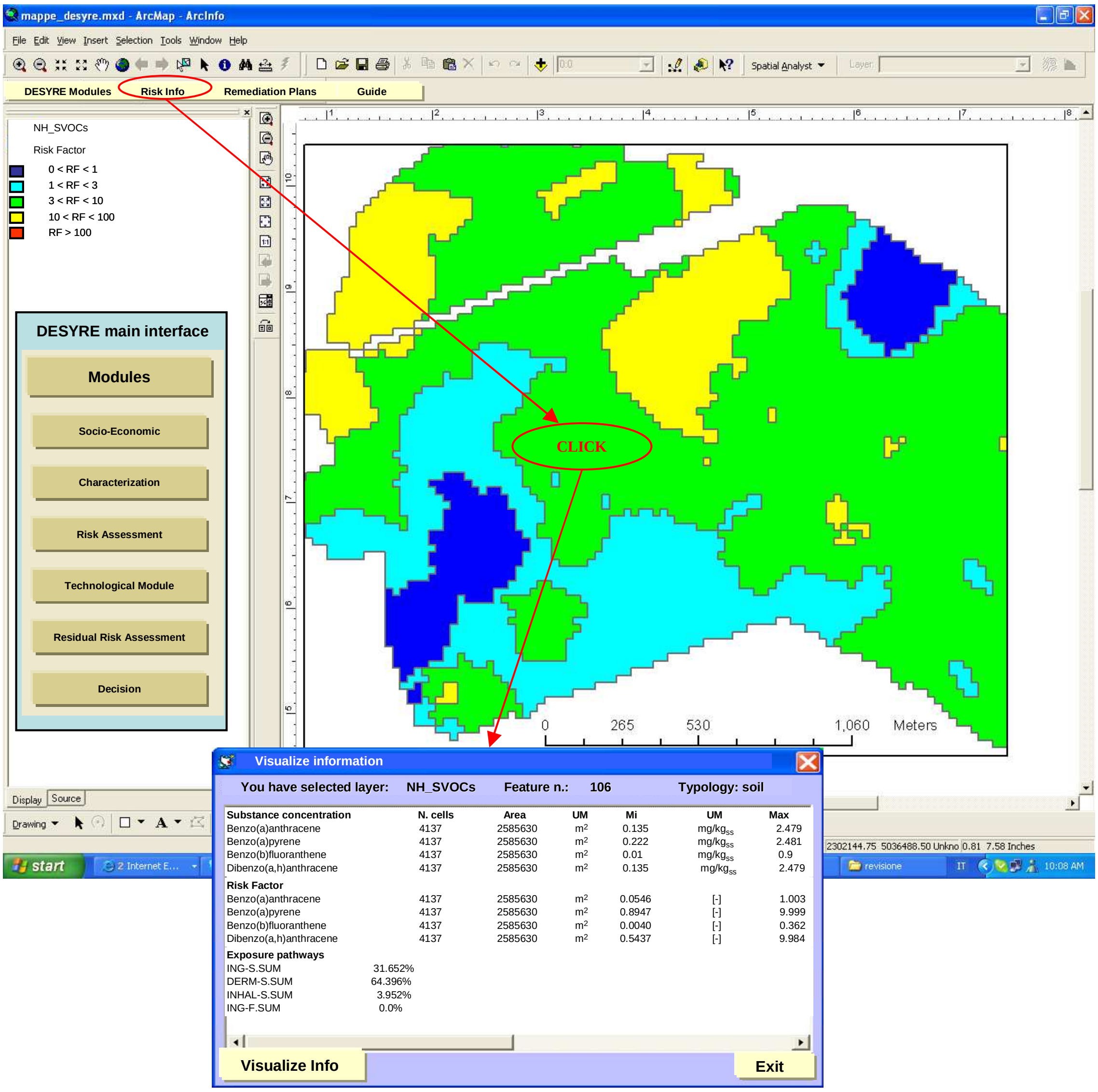


Figure 6.1 DESYRE software interface reporting the vector map of the the Maximum Risk Factor (RFmax) for the NH_SVOCs category. The table at the bottom shows the different information that can be obtained by clicking on the vector sub-areas of the risk factor map.

Among the advantages of vector maps it is the possibility to retrieve a more comprehensive characterization of the risk factor values; simply by clicking on the vector objects, the user of

DESYRE can visualize the associated tables with data about surface, contaminants concentration and contribution of various pathways to the overall exposure, as reported in Figure 6.1.

In Figure 3 the yellow zones indicate the highest maximum risk factor (RFmax) (10<RF<100) observed at the site. The most of the site show risk factors slightly higher than acceptability (3<RF<10). Two blue zones South and North of the site show Risk Factors within the acceptable range (RF<1) surrounded by a buffer zone with RF slightly exceeding the limit (1<RF<3). For the sake of example, Table 6.6 reports the information that can be recalled by clicking on the green zone. According to the results in the Table 6.6, the only contaminant posing a significant risk is Benzo(a)pyrene. However, also in the case of Benzo(a)pyrene the estimated risk factor exceed the acceptable risk factor (1) by a factor of 3, which is not a result of real concern due to the very conservative nature of the screening risk assessment.

As expected for low volatile substances, the most relevant exposure route appears to be the Soil Dermal Contact route (60.2% contribution to the overall exposure) followed by the Soil Ingestion route (29.6% contribution).

Table 6.6. Risk characterization of the “green” zone (3<RF<10) for NH-VOCs (Figure 6.1) in terms of total surface, concentration in soil (minimum, maximum, mean standard deviation), Risk factor (minimum, maximum, mean standard deviation), percentage contributes of different exposure pathways [Soil Ingestion (Ing_S), Dermal Contact with Soil (Dermal_C_S), Inhalation of volatile substances and particulate emission from soil (Inhal_S) and Ingestion of contaminated groundwater by soil percolation (Ing_Gw)] to the overall exposure.

	Surface	Concentration in soil				Risk Factor			
	m^2	mg/kgss	mg/kgss	mg/kgss	mg/kgss				
NH-SVOC		min	max	mean	StD	min	max	mean	StD
Benzo(a)anthracene	3125	0.6	0.6	0.6	0	0.3	0.3	0.3	0
Benzo(a)pyrene	3125	0.7	0.7	0.7	0	3.0	3.0	3.0	0
Benzo(b)fluorantene	3125	0.1	0.1	0.1	0	0	0	0	0
Dibenzo(a,h)anthracene	3125	0.6	0.6	0.6	0	0	0	0	0

Exposure routes contribution to the overall exposure	
Ing_S	29.6%
Dermal_C_S	60.2%
Inhal_S	4.1%
Ing_Gw	6.1%

Compared to NH-SVOCs, the inorganic chemicals appear to be an higher priority at the site, since the vector risk factor map (Figure 6.2) shows a widespread Risk Factor higher than 100 (red zone).

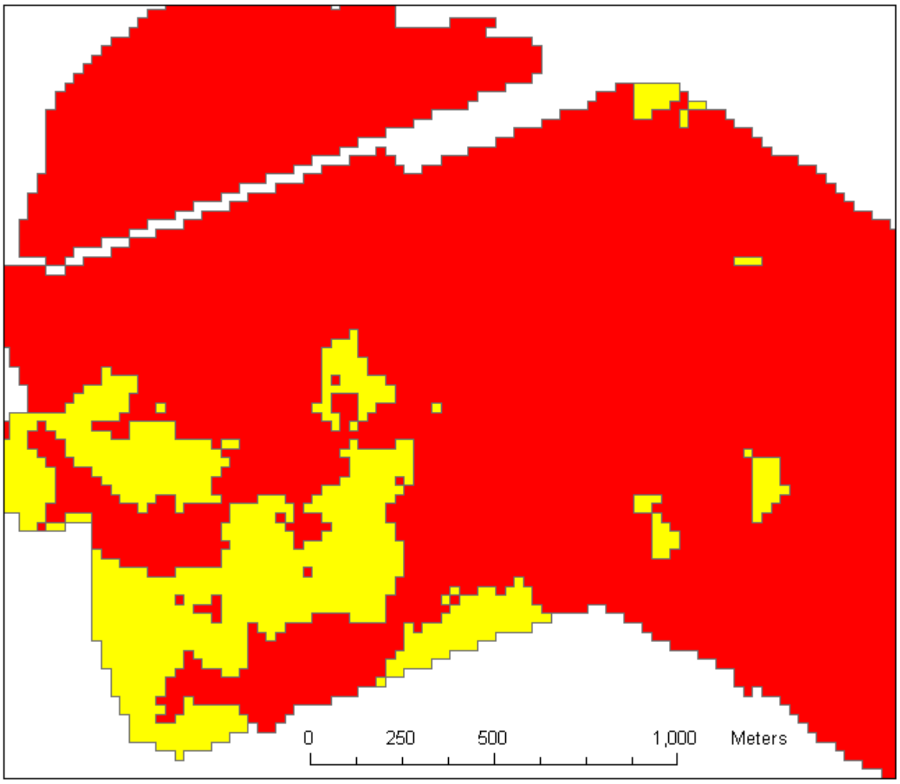


Figure 6.2. Vector map of the Maximum Risk Factor (RFmax) for the Inorganics category. RF <1, blue color; 1 < RF ≤ 3, cyan color; 3 < RF ≤ 10, green color; 10 < RF ≤ 100, yellow color; RF > 100, red color.

Further information on the red zone is reported in Table 6.7.

Table 6.7. Risk characterization of the "red" zone (>100) for inorganics (Figure 6.2) in terms of total surface, concentration in soil (minimum, maximum, mean standard deviation), Risk factor (minimum, maximum, mean standard deviation), percentage contributes of different exposure pathways [Soil Ingestion (Ing_S), Dermal Contact with Soil (Dermal_C_S), Inhalation of volatile substances and particulate emission from soil (Inhal_S) and Ingestion of contaminated groundwater by soil percolation (Ing_Gw)] to the overall exposure.

	Surface	Concentration in soil				Risk Factor			
	m^2	mg/kgss	mg/kgss	mg/kgss	mg/kgss				
Inorganics		min	max	mean	StD	min	max	mean	StD
Arsenic	4095630	8.2E+00	3.5E+02	5.7E+01	3.6E+01	3.2E+01	1.4E+03	2.2E+02	1.4E+02
Cadmium	4095630	3.0E-01	3.7E+02	2.5E+01	4.7E+01	1.0E-01	1.2E+02	8.0E+00	1.5E+01
Chromium	4095630	1.5E+01	5.1E+02	9.0E+01	8.2E+01	3.2E+01	1.1E+03	2.0E+02	1.8E+02
Manganese	4095630	1.9E+02	1.5E+03	6.7E+02	4.1E+02	1.9E+00	1.3E+01	6.7E+00	4.1E+00
Mercury	4095630	2.0E-01	1.2E+02	7.7E+00	1.0E+01	0.0E+00	4.9E+00	3.0E-01	4.0E-01
Lead	4095630	2.7E+01	7.6E+03	4.2E+02	6.4E+02	0.0E+00	2.4E+00	1.0E-01	2.0E-01
Copper	4095630	1.5E+01	7.2E+02	1.3E+02	9.6E+01	0.0E+00	0.0E+00	0.0E+00	0.0E+00
Vanadium	4095630	1.3E+01	5.3E+02	7.4E+01	5.3E+01	0.0E+00	3.0E-01	4.0E-02	3.0E-02
Zinc	4095630	1.9E+01	1.6E+04	1.5E+03	2.0E+03	0.0E+00	4.0E-01	3.0E-02	4.0E-02

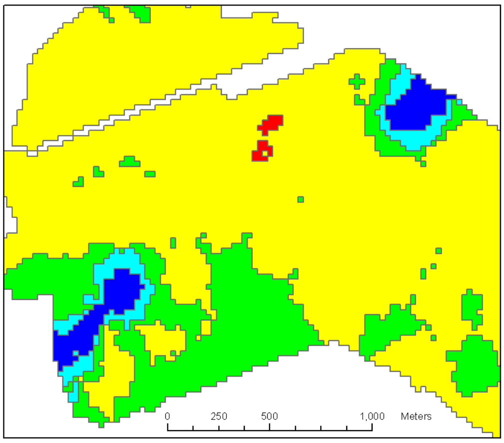
Exposure routes contribution to the overall exposure	
Ing_S	3.5%
Dermal_C_S	0.6%
Inhal_S	80.4%
Ing_Gw	15.5%

According to Table 6.7, arsenic and chromium both exceed the acceptable risk factor (1) by two orders of magnitude (RF>100). Cadmium shows maximum Risk Factor higher than 100 but lower mean values, indicating isolated hotspots with high concentrations. Manganese, mercury and lead have maximum risk factor values higher than 1, while copper vanadium and zinc have acceptable risk factor values in all the selected area. For the Inorganics category the exposure routes that more contribute to the multi-pathway Risk Factor is inhalation of volatile substances and particulate emission from soil (80%). It should be noted that since all these contaminants, with the exception of the mercury, are considered to be non volatile, the inhalation exposure route only represent the inhalation of soil suspended particulate.

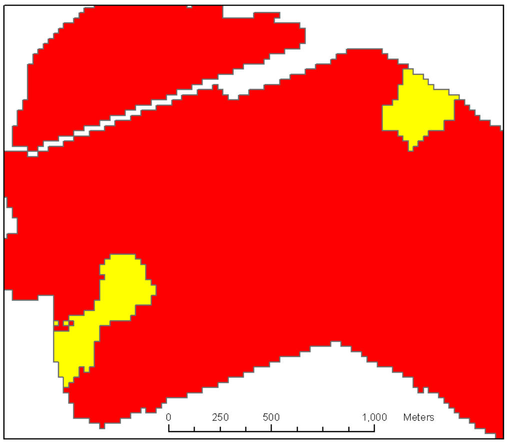
6.2.2 Probabilistic risk estimation

In the probabilistic approach, the Multi-pathway Acceptable Concentrations in Soil, MACSs, were calculated in the form of 50th percentile (P50) and 5th percentile (P5) according to Equation 4.6 and 4.7. For each CoC, the probabilistic approach generated two series of Risk Factor (RF) maps, the P50 and P95 RF maps, according to Equation 4.8 and 4.9, respectively.

The P50 and P95 RF maps for NH-SVOCs in the case study are reported in Figure 6.3, where maximum RF values within the category were considered (in alternative the sum of RF for each chemical within the category could be considered). The DESYRE software allows the user to define probability distribution inputs for anyone of the parameters. In the case study, all input parameters were defined in terms of probability distribution, as it is reported in Tables 6.2, 6.3, and 6.4.



a) Vector Map of the 50th percentile of the Maximum Risk Factor (RFmax).



b) Vector Map of the 95th percentile of the Maximum Risk Factor (RFmax).

Figure 6.3. Vector Maps of the 50th percentile (Figure a) and 95th percentile (Figure b) of the Maximum Risk Factor (RFmax) posed by NH_SVOCs soil contaminants. RF <1, blue color; 1 < RF ≤ 3, cyan color; 3 < RF ≤ 10, green color; 10 < RF ≤ 100, yellow color; RF > 100, red color.

Analogously to the deterministic output maps, DESYRE allows to interrogate the 50PC and 95PC maps for the CoCs concentration, RF statistics and contribution of exposure pathways.

The comparison of PC50 and PC95 maps provides an indicator of the uncertainty associated to RF estimates. Since PC95 shows RF estimates higher than 100 for the main part of the site, the decision maker should be aware of the very preliminary level of the characterization of the site and of the over conservatism of risk estimation thereof. Moreover, a rule of thumb is that higher the uncertainty estimation of the final outcome more the input parameters for which uncertainty is considered, and in particular if uncertainty associated to toxicological parameters is included, as it was made in the case study. In the scope of guiding further investigations at the site, a limited number of parameters can be treated in probabilistic ways based on their sensitivity to risk outputs (Nadal et al., 2001) and the actual susceptibility to be further investigated to reduce their uncertainty. For many parameters, out of which the toxicological values, further analyses can be technically or economically not viable in the scope of the remediation of the site, and for this reason they are usually defined by "conventional" values (commonly accepted as representative and reasonably conservative of general cases). Out of the input parameters, the concentration estimates of contaminants in soil are usually associated to a relevant uncertainty due to their spatial variability and limited sampling. Indicators of the uncertainty of concentration estimates can be derived from geostatistic interpolation methods, and in particular the sequential simulation method (Leonte and Schofield, 1996; Carlon et al., 2000; Gay and Korre, 2006). It should be borne in mind that the uncertainty of interpolation estimates depends on the spatial correlation of the predicted point with the sampling point, (Isaaks and Srivastava, 1989), and in the case of regular grid sampling the uncertainty is expected to be approx. constant across the site and increasing at the edge of the sampled area. For this reason, the uncertainty analysis of interpolation estimates can be a useful support to the planning of further investigations in the case of irregular sampling, while it does not provide valuable information in case of regular sampling. Since in the case study the soil sampling was performed according to a regular squared grid, the uncertainty associated to concentration estimates was not considered.

CHAPTER 7
SYRIADE application to the Poland case study

The SDSS has been tested on a sub-area of the Upper Silesia region in Poland which was selected since it is an acknowledged hot spot area in the context of environmental impacts from multi-source environmental contaminants from industrial activities. Moreover, the massive potential occurrence in the area of contaminated sites, and the presence of all the identified regional targets (i.e. humans, protected areas, surface water and groundwater) made of this area an appropriate application example for the proposed regional risk assessment methodology and its thorough evaluation.
The sub set of data used for the application was extrapolated from the real geodatabase collecting all the spatially resolved relevant available information of the region of concern which was developed by GIS experts from IETU (The Polish Institute for Ecology of Industrial Areas) in ArcGIS 9.2., using also data provided by the Polish Geological Institute (PGI). Moreover, the massive potential occurrence in the area of contaminated sites, and the presence of all the identified regional targets (e.g. humans, protected areas and so on) made of this area an appropriate application example for the proposed regional risk assessment methodology and its thorough evaluation.
The application of the developed regional risk assessment methodology follows the methodological approach presented in Chapter 5 and is divided the following steps:

- hazard analysis;
- vulnerability analysis;
- pathway analysis;
- risk factor estimation
- overall risk estimation (direct contact risk and regional risk);
- NUTS regional risk estimation.

7.1 Hazard analysis

The 10 potentially contaminated sites identified in the area of interest (within the Polish region) characterized by the related potentially polluting activities, the time of operation and the area of the site, are reported in Table 7.1. The related production activities are 13, meaning that more than one potentially polluting activity may have operated at the same site. The evaluation of the NACE code shows that the potentially polluting activities belong to the following economic sectors: Extraction of crude petroleum and natural gas (60), Manufacture of fabricated metal products, except machinery and equipment (25), Manufacture of chemicals and chemical products (20), Manufacture of basic pharmaceutical products and pharmaceutical preparations (21), Manufacture of basic metals (24) and Crop and animal production, hunting and related service activities (1).

Due to the available data about the sites in the original Polish dataset, the size parameter was calculated in this application by taking into consideration the area of the site.

The hazard of the potentially contaminated sites, where different potentially polluting activities are or have been present is estimated in dependence to the three main parameters described in Paragraph 5.1.3: toxicity, size and time.

Following the application procedure, described in Paragraph Paragraph 5.1.3 each hazard parameter value was normalized as continuous parameter and then the aggregation function was applied in order to estimate the hazard score for each activity. For each site characterized by more than one activity the hazard site score was obtained by summing each activity score.

The input data used for the hazard estimation and the final hazard score are reported in Table 7.1 for each site considered in the case study application. The spatial distribution of the potentially contaminated sites and their hazard score are reported in Figure 7.1.

Table 7.1. Input data used for the hazard score estimation and final hazard score. Sites are divided based on the administrative unit in which they are located. When site-specific data on operation time are not available (NA), the attributed default value is 10. (AUC (Administrative Unite Code), SITE ID and Activities ID = case-study application codes; NACE = Code number, according to the classification introduced by Council Regulation 3037/90/EEC; HRP = Hazard Ranking Position).

AUC	SITE ID	Activities ID	NACE	Number of contaminants categories per Risk phrases scores								Toxicity	Year of starting activities	Year of ending activities	Time norm	Size (area) (m^2)	Size norm	Hazard score for single activity	Hazard of the site	HRP
				5	10	15	20	25	30	35	40									
HH1	S01	101	60.24				2	6	2		6	6,7	NA	NA	10,0	6493	0,5	34,2	34,2	8
HH2	S02	102	20.51				1	6	4		4	6,2	NA	NA	10,0	141253	4,9	299,1	299,1	2
	S03	103	60.00				2	6	2		6	6,7	NA	NA	10,0	50245	3,3	221,8	221,8	4
	S04	104	24,50				1	3	4		6	6,2	1945	2006	6,1	7455	0,6	24,1	24,5	9
HH3	S05	105	21,00				2	6	4		7	8,1	NA	NA	10,0	46071	3,1	252,2	252,2	3
	S06	106	1,20					3				1,0	NA	NA	10,0	30237	2,4	24,6	185,5	5
		107	60,00				2	6	2		6	6,7	NA	NA	10,0	30237	2,4	160,9		
	S07	108	20,00				1	6	4		4	6,2	NA	NA	10,0	11814	1,3	77,3	77,3	7
	S08	109	25.23				1	4	4		7	7,1	1960	2006	4,6	1167196	7,8	221,3	1361,1	1
		110	25.22				1	4	4		7	7,1	1871	2006	10,0	1167196	7,8	553,3		
		111	25.21				1	4	4		7	7,1	1871	2006	10,0	1167196	7,8	553,3		
HH4	S09	112	60.24				2	6	2		6	6,7	NA	NA	10,0	15092	1,7	114,4	114,4	6
	S10	113	60,00				2	6	2		6	6,7	NA	NA	10,0	4876	0,3	19,6	19,6	10

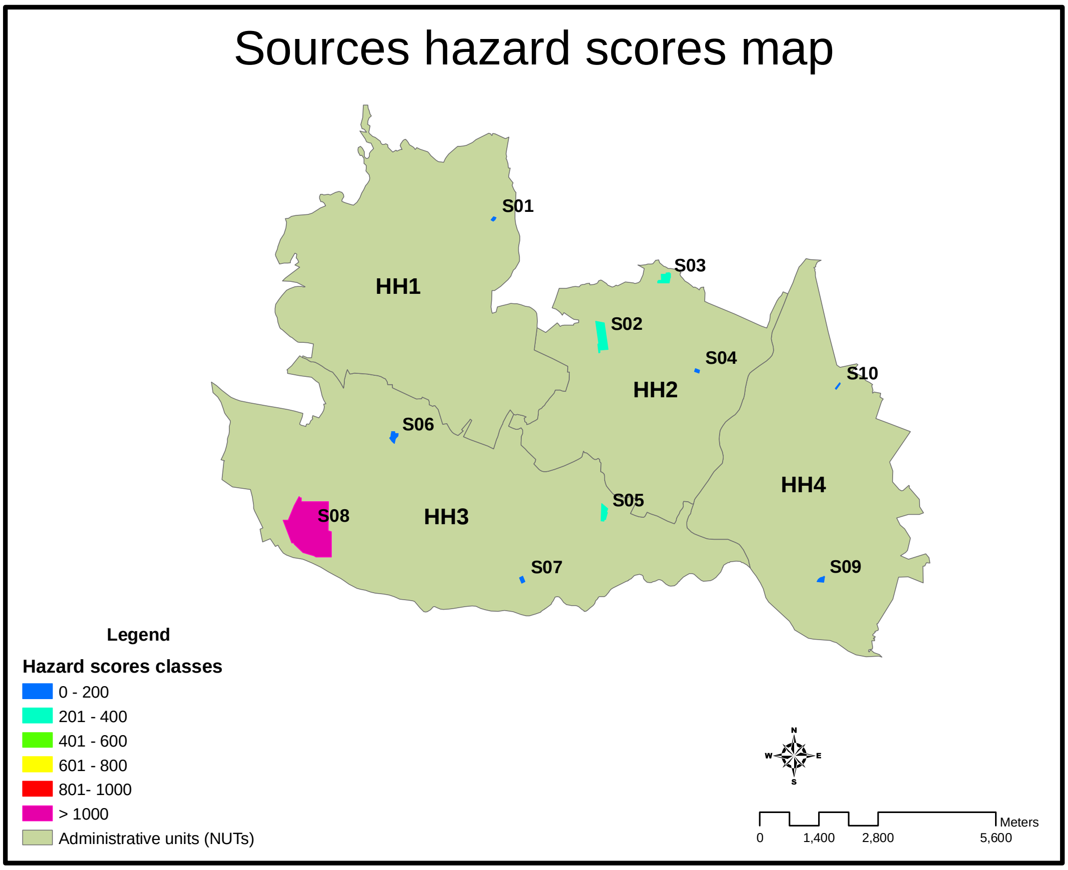


Figure 7.1. Results of the hazard analysis of potentially contaminated sites applied to the selected case study.

As reported in Table 7.1 the higher toxicity values were estimated for the sites S05 (Tox = 8.1) and S08 (Tox = 7.1). These sites are characterised by the highest number of contaminants categories (n=7) with the highest Risk Phrases Scores (FRS = 40) and a considerable number of contaminants categories in the other risk phrases classes. However, considering also the size and time criteria the final ranking position gives the first place to S08 (score = 1361,1). This high hazard score is due to the presence of three industrial activities on the same site which are all characterized by high toxicity values, high size values and high time values. S02 (score = 299,1) gains the second hazard ranking position even if the hazard score value is much lower than S08 and more similar to the other sites hazard scores as reported in Figure 7.1. In fact, most of the sites are in the second hazard scores class (scores between 200 and 400) or in the lower hazard scores class (score below 200).

7.2 Vulnerability analysis

According to the regional vulnerability methodology described in Paragraph 5.1.4 the first step of the vulnerability assessment is the regional target identification. In the selected area all identified regional targets, namely Human Health (HH), Surface Water (SW), Groundwater (GW), and Protected Areas (PA), were present and assessed. As required by the application, a preliminary analysis was performed on available dataset in order to be respondent to the application requirements. Specifically, the surface water layers were subjected to the spatial identification of homogenous surface water elements while the groundwater layers were subjected to the spatial identification of hydrogeological homogenous areas.

The identification of the homogeneous surface water elements was made through the use of the Watershed with Strahler Order 1 (WSO1) within the hierarchical river network defined at the European level (JRC, 2007): all the segment rivers belonging to the WSO1 catchment scale level compose a single homogeneous surface water element.

In order to define the hydro-geological homogenous areas, the Watershed with Strahler Order 3 (WSO3) (JRC, 2007) was used. The WSO3 catchments boundaries and the related rivers spatial distribution were both used for the definition of the hydro-geological homogenous areas boundaries according to the approach explained in the work of Hubbert (Hubbert, 1940). As far as Human Health and Protected Areas are concerned, the NUT 5 (Nomenclature of Territorial Units for Statistics) spatial extends and the protected areas shapes were used as spatial features for the vulnerability assessment. The spatial extent of each target is show in Figure 7.2.

For each regional target the list of vulnerability relevant parameters (attributes for MCDA) collected in step 2 was adapted to the case study according to the available information and reported, with related description and classes used in the assessment, in Table 7.2. The selected parameters represent meaningful characteristics of the regional receptors which influence the receptors vulnerability to contamination. For examples in the case of human health vulnerability assessment, the percentage of vulnerable groups represents the population percentage composed of children (< 14 years old) and elderly persons (older than 65) which result to be more strongly affected by the presence of contamination. The degree of isolation parameter for the ground water vulnerability assessment represents the intrinsic soil property which influences the capacity of the contamination to reach the groundwater level, while the typology parameter (surface water flow rate) expresses the surface water dilution capacity to contamination.

For both groundwater and surface water assessment, the quality class and the use class give an indication of the importance of the water resources for human activities. For each identified vulnerability attribute the related values were collected and the spatial attributions were performed (step 3 in Chapter 5). As reported in the second column of Table 7.2, for most of the parameters the spatial aggregation function used for the parameters values spatial attribution is the widest area function. For the remaining parameters the conservative values were selected through the use of the maximum value function applied to the normalized values.

Table 7.2. List of targets vulnerability relevant parameters with related description and classes used in the case study.

Human health

Parameter	Spatial aggregation function	Class	Description	Score
Population density (P.D.)	-	>= 2000 people/km2	High density	1
		1000 people/km2	Medium density	0,6
		250 people/km2	Low density	0,2
Land use (L.U.)	-	Residential	Urban fabric	1
		Agricultural/Livestock	Agricultural areas	0,8
		Park/School	Green urban areas and sport and leisure facilities	0,6
		Forest	Forest	0,4
		Industrial/Commercial	Industrial or commercial units	0,2
		Isolated area	Isolated area	0
Percentage of vulnerable groups (%V.G.)	-	> 40	High	1
		32-40	Medium-high	0,7
		22-31	Medium	0,5
		<22	Low	0,3

Ground water

Parameter	Spatial aggregation function	Class	Description	Score
Degree of isolation from soil surface (Iso.)	widest area	a – low or lacking	degree of isolation less than 15 m thick till or 5 thick clay in a overburden	1
		b – mid	degree of isolation between 15 m to 50 m thick till or between 5 to 10 m thick clay in a overburden	0,5
		c – high	degree of isolation more than 50 m thick till or more than 10 m thick clay in a overburden	0
Number of water bearing layers (B.L.)	widest area	1	one layer	0
		2	two layers	0,25
		3	three layers	0,5
		4	four layers	0,75
		> 5	more than five layers	1
Quality class (Qua.)	widest area	BRA	the main groundwater horizon do not exist/lack of information	1
		BR	the main groundwater horizon do not exist	0
		I a	water quality good and stable, do not require conditioning	1
		I b	water quality good, but can be unstable because of insufficient isolation layer, water do not require conditioning	0,75
		II a	water quality moderate, water require simple sanitation, (according to procedures used in standard water conditioning station in water supply systems: removal of iron, manganese, aeration), the quality is stable	0,5
		II b	water quality moderate, water require simple sanitation (according to procedures used in standard water conditioning stations in water supply systems: removal of iron, manganese, aeration), the quality is potentially unstable because of insufficient isolation layer	0,25
		III	water quality bad, water require complicated sanitation (according to procedures beyond those used in the standard station of water conditioning in the water supply system)	0
Use class (Use)	more conservative value (maximum normalised value)	Municipal use		1
		Private well	use not specified	0,9
		Irrigation	food, vegetables	0,85
		Other use not specified	Golf, parks, no food, vegetables, etc.	0,55
		Industrial use		0,3
		Non used		0,1
		No groundwater body present		0

Protected areas

Parameter	Spatial aggregation function	Class	Description	Score
Protection typology (Typ.)	-	State	Protected areas at state level	1
		Region	Protected areas at regional level	0,8
		Natura 2000	Sites identified by Natura 2000 database not included in the first two classes	0,6
		Ecological areas	Ecological area protected at local level	0,5
		Biogenic monuments	Biogenic natural monuments protected at local level	0,4
		Abiogenic monuments	Abiogenic natural monuments protected at local level	0,3
		Cultural heritage	Cultural heritage monuments protected at local level	0,2
Extension (Ext.)	-	>= 2000000 m2	Huge extension	1
		1500000 m2	Large extension	0,83
		500000 m2	Medium extension	0,67
		50000 m2	Small extension	0,3
		5000 m2	Tiny extension	0,03

Surface water

Parameter	Spatial aggregation function	Class	Description	Score
Quality class (Qua.)	widest area	Water I class (I)	point on the stream, water body or its part, where was determined that the water meets drinking water standards, food industry needs or other industries requiring drinking water quality and the quality for living conditions of Salmonidae species	1
		Water II class (II)	point on the stream, water body or its part where it was determined that the water is usable for living conditions for farming fish other than Salmonidae species, husbandry use, bathing activities and recreational activities.	0,6
		Water III class (III)	point on the stream, water body or its part, where it was determined, that the water is usable for industry purposes except for the industries requiring drinking water quality, watering purposes in agriculture and horticulture purposes.	0,3
		Water non class (N)	point on the stream, water body or its part, where was determined that the water is not usable in any of the use categories because of its above limits contamination.	0
Typology (Typ.) surface water flow rate (m3/s)	more conservative value (maximum normalised value)	0	Small to moderate stream	1
		10	Moderate to large stream	0,8
		30	Large stream or river	0,6
		100	Large river, coastal tidal waters, shallow ocean zone, great lake	0,5
		1000	Very large river, moderate ocean zone	0,3
		>= 2000	Extreamely large river, ocean zone	0
Use class (Use)	more conservative value (maximum normalised value)	Potable		1
		Recreational (Swimming/Bathing)		0,8
		Piscicolture (salmonid/Shellfish)		0,75
		Irrigation (Food crop)		0,65
		Piscicolture (Cyprinid)		0,6
		Irrigation (No food crop)		0,55
		Industrial/Other		0,3
		Non used		0,2

According to the proposed methodology, the 4th step concerns the normalisation of the vulnerability attribute values. Normalization functions were performed applying the values reported in Table 7.2, and normalization results can be found in Table 7.4 (i.e., columns whose names contain the "norm." postfix) together with their generating original values. In a precautionary approach the normalized values corresponding to missing original values were set equal to 1.

In the 5th and final step, normalized values are used as input data to be aggregated into the final vulnerability score by Choquet integral application. The non monotonic measures used as Choquet integral parameters are reported in Table 7.3. In the case study application, two experts from IETU (The Polish Institute for Ecology of Industrial Areas) acted as one environmental expert during the MCDA questionnaires filling. It should be noted that in some cases monotonicity has been violated (highlighted rows): this means that some adversarial phenomena between attributes are present. By examining the questionnaires in can be noted that some criteria have been considered more important than others. In particular such prevalence of importance of few criteria among others is the cause of the non monotonic behavior of scores. This happens, for example, in the Human Health table where the percentage of vulnerable groups criterion results to be the most important parameter.

After attributes aggregation, final vulnerability scores were obtained for each of the four different targets and reported in Table 7.4; while in Figure 7.3 the related vulnerability score maps are showed.

Table 7.3. Non monotonic measures questionnaires filled in by the expert who has to evaluate the different combinations where the parameters may be in the highest class (1= best value) or in the lowest class (0= worse value). The attributed scores are in the 0-100 range.

Human health			
P.D.	L.U.	%V.G.	Score
0	0	0	0
0	0	1	85
0	1	0	40
1	0	0	50
0	1	1	90
1	0	1	95
1	1	0	60
1	1	1	100

Protected area		
Typ.	Ext.	Score
0	0	80
0	1	80
1	0	100
1	1	100

Ground water				
Iso.	B.L.	Qua.	Use	Score
0	0	0	0	5
0	0	0	1	45
0	0	1	0	15
0	1	0	0	5
1	0	0	0	65
0	0	1	1	50
0	1	0	1	25
1	0	0	1	95
0	1	1	0	10
1	0	1	0	70
1	1	0	0	55
0	1	1	1	30
1	0	1	1	100
1	1	0	1	75
1	1	1	0	60
1	1	1	1	80

Surface water			
Qua.	Typ.	Use	Score
0	0	0	0
0	0	1	30
0	1	0	50
1	0	0	10
0	1	1	90
1	0	1	40
1	1	0	70
1	1	1	100

Table 7.4. Vulnerability scores estimations trough the integration of the normalised attributes values by means of the Choquet integral.

Human health							
ID	Population density	Population density norm.	Land use	Land use norm.	Percentage of vulnerable groups	Percentage of vulnerable groups norm.	Vulnerability
HH01	1607	0,84	residential area	1,00	29%	0,44	**0,74**
HH02	1607	0,84	residential area	1,00	29%	0,44	**0,74**
HH03	1607	0,84	residential area	1,00	26%	0,38	**0,72**
HH04	382	0,27	agricultural area	0,80	28%	0,42	**0,56**

Protected areas					
ID	Protection typology	Protection typology norm.	Extension	Etension norm.	Vulnerability
PA01	Cultural heritage	0,20	380	0,00	**0,20**
PA02	Cultural heritage	0,20	380	0,00	**0,20**
PA03	Cultural heritage	0,20	380	0,00	**0,20**
PA04	Cultural heritage	0,20	380	0,00	**0,20**
PA05	Cultural heritage	0,20	380	0,00	**0,20**
PA06	Cultural heritage	0,20	380	0,00	**0,20**
PA07	Cultural heritage	0,20	380	0,00	**0,20**
PA08	Cultural heritage	0,20	380	0,00	**0,20**
PA09	Cultural heritage	0,20	380	0,00	**0,20**
PA10	Cultural heritage	0,20	380	0,00	**0,20**
PA11	Cultural heritage	0,20	380	0,00	**0,20**
PA12	Cultural heritage	0,20	380	0,00	**0,20**
PA13	Cultural heritage	0,20	380	0,00	**0,20**
PA14	Biogenic monuments	0,40	380	0,00	**0,40**
PA15	Biogenic monuments	0,40	380	0,00	**0,40**
PA16	Biogenic monuments	0,40	380	0,00	**0,40**
PA17	Biogenic monuments	0,40	380	0,00	**0,40**
PA18	Biogenic monuments	0,40	380	0,00	**0,40**
PA19	Biogenic monuments	0,40	185847	0,41	**0,41**

Ground water									
ID	Degree of isolation from soil surface	Degree of isolation from soil surface norm.	Number of water bearing layers	Number of water bearing layers norm.	Quality class	Quality class norm.	Use class	Use class norm.	Vulnerability
GW01	c	0,00	1	0,00	BR	0,00		1,00	**0,45**
GW02	a	1,00	3	0,50	BR	0,00		1,00	**0,85**
GW03	a	1,00	2	0,25	BR	0,00	p	0,30	**0,69**
GW04	b	0,50	2	0,25	II b	0,25	p	0,30	**0,38**
GW05	b	0,50	3	0,50	II b	0,25	p	0,30	**0,35**
GW06	b	0,50	3	0,50	II b	0,25	k	1,00	**0,61**
GW07	a	1,00	2	0,25	II b	0,25	k	1,00	**0,91**
GW08	a	1,00	3	0,50	II b	0,25	k	1,00	**0,86**
GW09	b	0,50	3	0,50	BR	0,00		1,00	**0,60**

Surface water							
ID	Quality class	Quality class norm.	Typology	Typology norm.	Use class	Use class norm.	Vulnerability
SW01	N	0,00	0,11	1,00	R	0,80	**0,66**
SW02	N	0,00	0,13	1,00		1,00	**0,70**
SW03	N	0,00	0,06	1,00		1,00	**0,70**
SW04	N	0,00		1,00	I	0,30	**0,56**
SW05	N	0,00		1,00		1,00	**0,70**
SW06	N	0,00	0,04	1,00		1,00	**0,70**
SW07	N	0,00		1,00	NFI	0,55	**0,61**
SW08	N	0,00	1,03	0,98		1,00	**0,69**
SW09	N	0,00		1,00		1,00	**0,70**
SW10	N	0,00		1,00		1,00	**0,70**
SW11	N	0,00		1,00		1,00	**0,70**
SW12	N	0,00	1,42	1,00		1,00	**0,68**
SW13	N	0,00		1,00	I	0,30	**0,56**
SW14	N	0,00	0,84	0,98		1,00	**0,69**
SW15	N	0,00		1,00		1,00	**0,70**
SW16	N	0,00		1,00	I	0,30	**0,56**
SW17	N	0,00		1,00	I	0,30	**0,56**
SW18	N	0,00		1,00		1,00	**0,70**
SW19	N	0,00		1,00		1,00	**0,70**
SW20	N	0,00		1,00	NFI	0,55	**0,61**
SW21	N	0,00		1,00		1,00	**0,70**
SW22	III	0,30		1,00		1,00	**0,79**
SW23	N	0,00		1,00		1,00	**0,70**

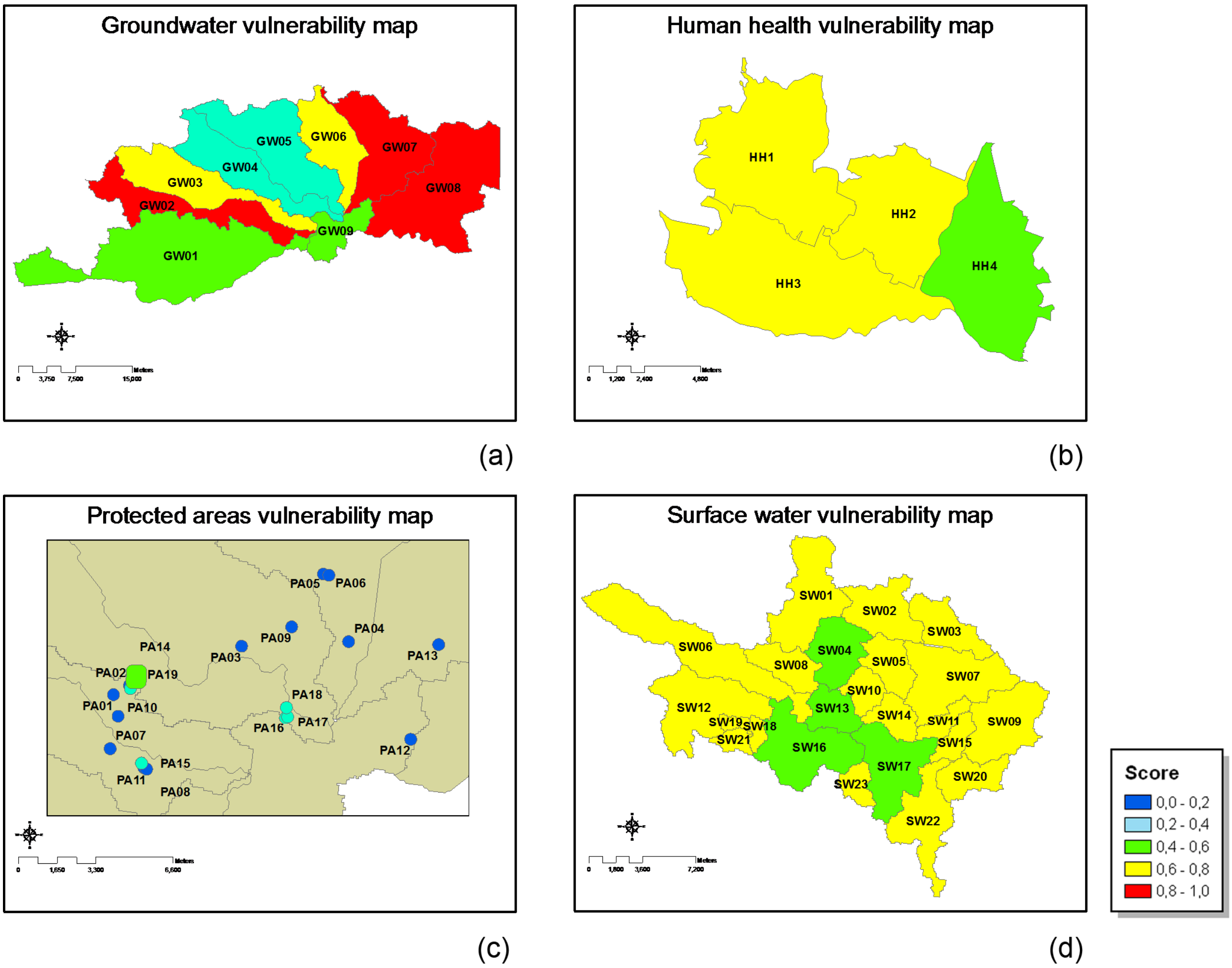


Figure 7.2: Vulnerability to contaminated sites scores maps: a) Groundwater homogeneous areas vulnerability scores, b) NUTS vulnerability scores, c) Protected Areas vulnerability scores and d) homogenous Surface Water elements vulnerability scores. Protected areas were represented in the geodatabase by point features with small extensions, which were widened by the application of a buffer only for visualization purpose.

As far as groundwater is concerned, a wide area representing 2 different groundwater homogeneous areas of higher vulnerability is located on the right side of map (a) in Figure 7.2, while the central area of the same map identifies two groundwater homogeneous areas where the vulnerability to contamination is lower. In the case of Human Health (Fig 7.2b) the most vulnerable NUTs are those with high population density, higher percentage of vulnerable groups and characterized by a residential land use. On the other hand the agricultural area with low population density received a lower vulnerability score.

As far as protected areas are concerned (Fig 7.2c), the vulnerability score identified three main priority levels: the most vulnerable protected area is the bigger biogenic monument (i.e., PA19); the second priority level is represented by others biogenic monuments; while the lower priority level is represented by cultural heritage monuments, since they are supposed to be less sensitive to contamination occurrence. Moreover, it has to be underlined that all the selected protected areas, with only one exception, have the same extension since they were represented in the geodatabase by point features.

Finally, the surface water vulnerability assessment evidences a quite homogeneous vulnerability scores distribution with a central area where the vulnerability is lower (Fig 7.2d). This homogeneous vulnerability distribution is caused by the case study dataset in which some original values were missing and for which, in a precautionary approach, the corresponding normalized values were set equal to 1, producing the higher vulnerability scores.

7.3 Pathway analysis

According to Paragraph 5.1.5, three regional pathways (i.e. groundwater, surface water and wind pathway) were identified, linking the potentially contaminated sites with the protected areas. For these pathways, a relevance pathways analysis was performed in order to estimate the associated scores. The first step of the relevance pathways analysis is the identification of the parameters (called attributes in MCDA) relevant for the estimation of the pathway relevance. Then the spatial attribution of the identified pathway parameter values was performed applying the spatial aggregation functions described in Paragraph 5.1.4.3. Once each identified pathway spatial entity (i.e. surface water homogeneous elements and groundwater homogeneous areas) had its own pathway parameter value for each identified parameter, a normalisation function was applied according to the methodological explanation reported in Paragraph 5.1.4.4. The identified pathway parameters, the spatial aggregation function and the normalisation function class values are reported in Table 7.5.

Table 7.5. Identified pathway parameters, spatial aggregation function used for each parameter and normalization function class values.

Groundwater				
Parameter	Spatial aggregation function	Class	Description	Score
Degree of isolation from soil surface (DoI)	widest	a – low or lacking	degree of isolation less than 15 m thick till or 5 thick clay in a overburden	1
		b – mid	degree of isolation between 15 m to 50 m thick till or between 5 to 10 m thick clay in a overburden	0,5
		c – high	degree of isolation more than 50 m thick till or more than 10 m thick clay in a overburden	0
Number of the water bearing layers (NoWBL)	widest	1		0
		2		0,25
		3		0,5
		4		0,75
		>5		1
Trasmissivity (T) (cm/s)	area based weighted average	> = 1500	high	1
		750	medium-high	1
		350	medium	0,75
		150	low - medium	0,5
		50	low	0,25
		0		0
Surface water				
Parameter		Class	Description	Score
Typology (Typ.) surface water flow rate (m3/s)	more conservative value (maximum normalized value)	0		1
		10	Small to moderate stream	0,8
		30	Moderate to large stream	0,6
		100	Large stream or river	0,5
		1000	Large river, coastal tidal waters, shallow ocean zone, great lake	0,3
		>= 2000	Very large river, moderate ocean zone	0

Finally, in the 5th step, normalized values are used as input data to be aggregated into the relevance pathway score through the Choquet integral application (Paragraph 5.1.4.5). The Choquet integral requires the definition of monotonic measures, which can be elicited by experts through the use of an ad-hoc questionnaire, as the one reported in Table 7.6 for the groundwater pathway. As far as the surface water pathway is concerned, only one parameter is used for the definition of the homogeneous surface water element relevance: the surface water typology, which normalized represent the homogeneous surface water element relevance score. Thus no questionnaire was developed for this pathway. The surface water pathway relevance score is reported in Table 7.7. It has to be underlined that the path which links the source with the receptor along the river can include more than one homogeneous surface water element, thus a suitable surface water pathway relevance score should be assigned to the whole path.

Finally, the wind pathway relevance analysis was not performed, since all the targets located inside the buffer of radius d are considered to be impacted in the same way and the pathway relevance score was considered to be equal to 1. Moreover, wind pathway implementation could consider the distance between sources and receptors. According to the above considerations, the Choquet integral was applied only to the groundwater pathway and the related questionnaire is reported in Table 7.6.

Table 7.6. Non monotonic measures questionnaires for the Groundwater pathway. An expert evaluated the score for each coalition of the questionnaire, where parameters may be in the highest class (1= highest pathway relevance) or in the lowest class (0= lowest pathway relevance). The assigned score can vary between 0 and 100, where 100 is assigned to identify the maximum pathway relevance and 0 the minimum.

Ground water pathway			
DoI	NoWBL	T	Score
0	0	0	0
1	0	0	50
0	1	0	0
0	0	1	60
1	1	0	55
1	0	1	100
0	1	1	45
1	1	1	90

Table 7.6 reports the Groundwater pathway measure based on the possible coalitions of Degree of isolation from soil surface (DoI), Number of the water bearing layers (NoWBL) and Trasmissivity (T). This means to create a map between the power set of the set of selected attributes (DoI, NoWBL and T) and the corresponding scores. The corresponding scores were obtained asking to the Polish environmental expert from IETU to fill in the questionnaire reported in Table 7.6.

In general, the main characteristic of non monotonic measures is that each attribute taken into account may be a benefit in some coalitions and a cost in others. This kind of measures is therefore suitable to represent both synergic and adversarial relations between attributes.

By examining Table 7.6, it can be stressed out that T plays an important role inside the measure. In fact, its presence alone influences 60% of the effect on the groundwater pathway. Moreover, T supported by DoI gives the maximum effect on the groundwater pathway (100%).

The final pathway relevance scores estimated by means of the Choquet integral are reported in Table 7.7.

Table 7.7. Groundwater and surface water pathways relevance scores.

Groundwater							
ID	Degree of isolation from soil surface	Degree of isolation from soil surface norm.	Number of water bearing layers	Number of water bearing layers norm.	Trasmissivity	Trasmissivity norm.	Pathway relevance score
GWp01	c	0,00	1	0,00	194	0,56	0,33
GWp02	a	1,00	3	0,50	299	0,69	0,79
GWp03	a	1,00	2	0,25	409	0,79	0,87
GWp04	b	0,50	2	0,25	256	0,63	0,55
GWp05	b	0,50	3	0,50	54	0,26	0,37
GWp06	b	0,50	3	0,50	124	0,43	0,43
GWp07	a	1,00	2	0,25	405	0,78	0,87
GWp08	a	1,00	3	0,50	314	0,71	0,80
GWp09	b	0,50	3	0,50	385	0,77	0,61

Surface water		
ID	Typology	Typology norm.
SWp01	0,11	1,00
SWp02	0,13	1,00
SWp03	0,06	1,00
SWp04		1,00
SWp05		1,00
SWp06	0,04	1,00
SWp07		1,00
SWp08	1,03	0,98
SWp09		1,00
SWp10		1,00
SWp11		1,00
SWp12	1,42	1,00
SWp13		1,00
SWp14	0,84	0,98
SWp15		1,00
SWp16		1,00
SWp17		1,00
SWp18		1,00
SWp19		1,00
SWp20		1,00
SWp21		1,00
SWp22		1,00
SWp23		1,00

Once the groundwater pathway score was estimated, receptors reached by the contamination were identified. To this end, the following spatial analysis were implemented:
identification of the sources and receptors which fall in the same hydrogeological homogenous areas;
attribution of the information concerning the groundwater table level to all sources and receptors;
for each source, identification of all the receptors (protected areas) falling in the same hydrogeological homogenous area and, at the same time, having a groundwater table level equal or lower than the one associated to the source.

The sources-receptor pairs for groundwater are reported in Table 7.8.

As far as the surface water is concerned, once the homogeneous surface water element relevance scores are estimated, the following step is the identification of the receptors reached by the contamination. To this end the following spatial analyses were implemented:

- the estimation of a buffer of 500 meters around the rivers;
- the identification of the sources and the receptors which fall within the buffers;
- the construction of a river graph which links all the surface water elements in order to identify the river flow direction;
- the association of the sources and the receptors to the river graph in order to identify all the possible existing links between the sources and the receptors;
- the estimation of the surface water pathway relevance, through the integration of the scores associated to each homogenous surface water element composing the pathway selecting the most conservative value (i.e. the higher).

The sources-receptor pairs for surface water are reported in Table 7.8.

Table 7.8. Regional risk assessment results. The grey columns contain the risk factors of each source with respect to suitable pathways and impacted receptors, in consideration of source hazard, pathway score and receptors vulnerability (Si= contamination source; GW and gw = groundwater; SW and sw = surface water; HH = Human Health; PA = Protected Area, NUTS = Nomenclature of Territorial Units for Statistics and RRP = Risk Ranking Position). The underlined column contains the estimated risk factors for the direct contact pathways (Human Health, Ground water and Surface water), the bold column contains the regional relative risk score of each source as sum of all the estimated risk factors, while the last column contains the NUTS relative risk as a sum of all the regional relative risk scores of the sites located in the analyzed NUTS.

SITE ID	NUTs	HH risk factor	GW ID	GW risk factor	SW ID	SW risk factor	ID of PA reached by WIND	PA WIND risk factor	ID of PA reached by SW	SW pathway ID	PA-SW: risk factor	ID of PA reached by GW	GW pathway ID	PA-GW: risk factor	Regional relative risk (direct contact)	**Regional relative risk**	RRP	NUTS risk
S01	HH1	25.3	GW06	20.9											46.2	**46.2**	8	46.2
S02	HH2	221.3	GW05	104.7			PA9	59.8				PA3 PA9	gw05	44.3	326.0	**430.1**	5	1115.2
S03		164.1	GW06	135.3	SW02	110.9	PA5 PA6	88.7	PA05 PA06	sw02	88.7	PA4 PA5 PA6	gw06	57.2	410.3	**645.0**	4	
S04		18.1	GW06	14.9			PA4	4.9				PA4	gw06	2.1	33.1	**40.1**	9	
S05	HH3	181.6	GW04	95.8	SW14	123.6	PA16 PA17 PA18	302.6	PA16 PA17 PA18	sw14	296.6	PA17 PA18 PA16	gw4	166.4	401.0	**1166.7**	2	9808.8
S06		133.6	GW04 GW05	135.4	SW13	103.9	PA19	76.1	PA03 PA16 PA17 PA18	sw13 sw10 sw14	259.7	PA16 PA17 PA18	gw4	122.4	372.9	**831.0**	3	
S07		55.7	GW03	53.3											109.0	**109.0**	7	
S08		980.0	GW03	939.2	SW16	762.2	PA10 PA07 PA01 PA14 PA02 PA19	2735.8	PA07 PA11 PA15	sw16	1088.9	PA1; PA2; PA10; PA19;	gw3	1196.0	2681.4	**7702.1**	1	
S09	HH4	64.1	GW08 GW09	167.0			PA12	22.9							231.1 -	**254.0**	6	286.7
S10		11.0	GW07	17.8			PA13	3.9							28.8	**32.7**	10	

7.4 Relative risk analysis

The final analysis of the relative regional risk assessment of potentially contaminated sites included four main steps:

- calculation of the risk factors of each source with respect to suitable pathways and impacted receptors, in consideration of source hazard, pathway score and receptors vulnerability as described in Paragraph 5.1.2;
- calculation of the relative risk score of each source as sum of all the estimated risk factors for all the direct contact pathways (Human Health, Ground water and Surface water)as described in Paragraph 5.1.6;
- calculation of the relative risk score of each source as sum of all the estimated risk factors all the analysed pathways (Human Health, Ground water, Surface water, Protected areas reached by the wind pathway, Protected Areas reached by the surface water pathway and Protected Areas reached by the groundwater pathway);
- calculation of the relative risk score for each NUTS as the sum of the regional relative risk scores estimated for each site located in the analyzed NUTS.

The relative regional risk analysis results are reported in Table 7.8, while the related risk factor maps are reported in Figures 7.3 to 7.10.

The regional risk assessment results are reported in Table 7.8, where the grey columns contain the risk factors of each source with respect to suitable pathways and impacted receptors, in consideration of source hazard, pathway score and receptors vulnerability. The 16th column contains the estimated risk factors for the direct contact pathways (Human Health, Ground water and Surface water), the 17th column contains the regional relative risk score of each source as sum of all the estimated risk factors, while the last column contains the NUTS relative risk as a sum of all the regional relative risk scores of the sites located in the same NUTS.

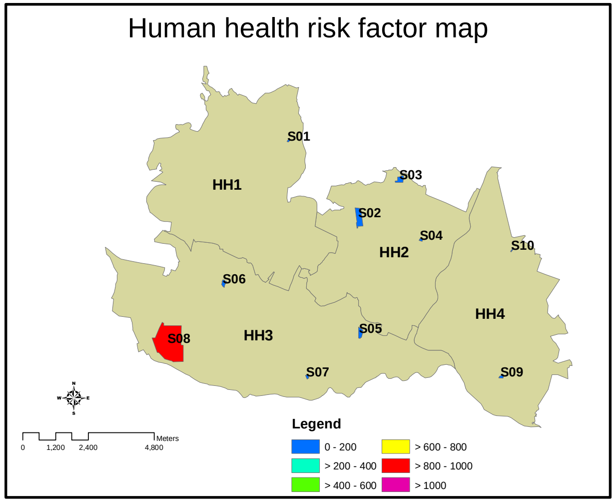


Figure 7.3. Results of the risk factor calculation for human health (HH) applied to the Upper Silesia region. Scores closer or higher than 1000 identify sites of higher risk factor.

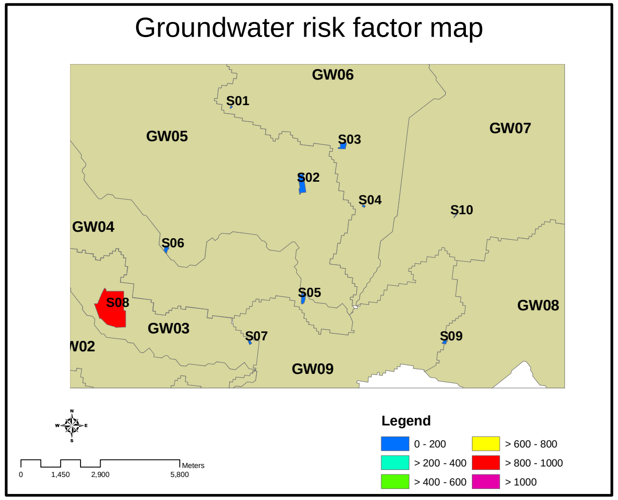


Figure 7.4. Results of the risk factor calculation for groundwater applied to the Upper Silesia region. Scores closer or higher than 1000 identify sites of higher risk factor.

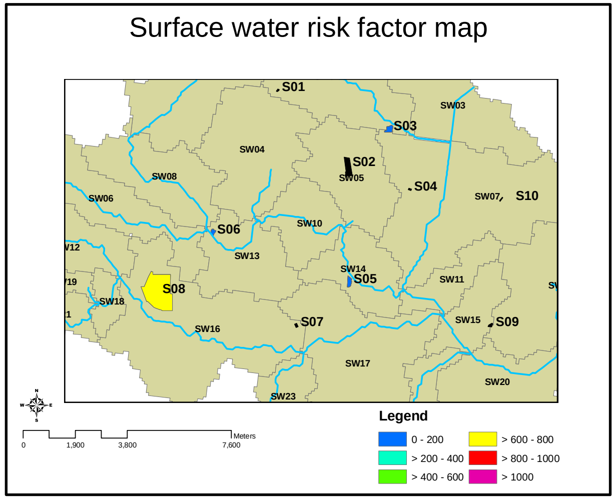


Figure 7.5. Results of the risk factor calculation for surface water applied to the Upper Silesia region. Scores closer or higher than 1000 identify sites of higher risk factor. The black sites are too far from the rivers to impact them.

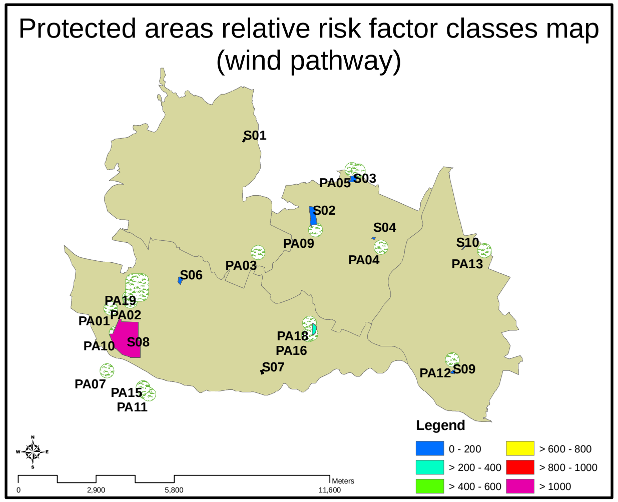


Figure 7.6 – Results of the risk factor calculation for protected areas impacted by wind transported contamination applied to the Upper Silesia region in the form of relative risk factor classes map. Scores closer or higher than 1000 identify sites of higher risk factor. The black sites are those which do not reach any protected areas.

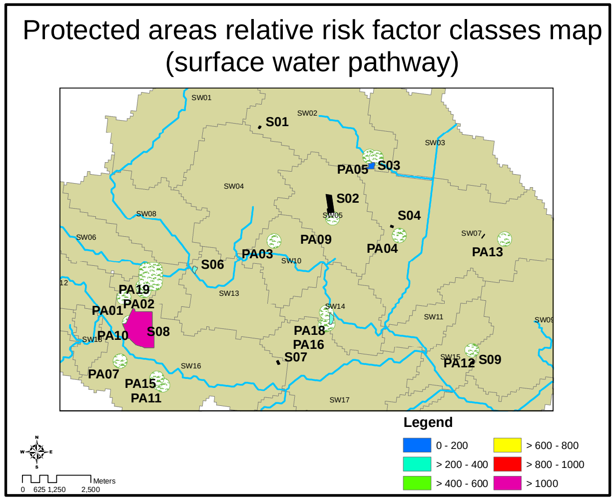


Figure 7.7 – Results of the risk factor calculation for protected areas impacted by the contaminated surface water applied to the Upper Silesia region in the form of relative risk factor classes map. Scores closer or higher than 1000 identify sites of higher risk factor. The black sites are those which do not reach any protected areas.

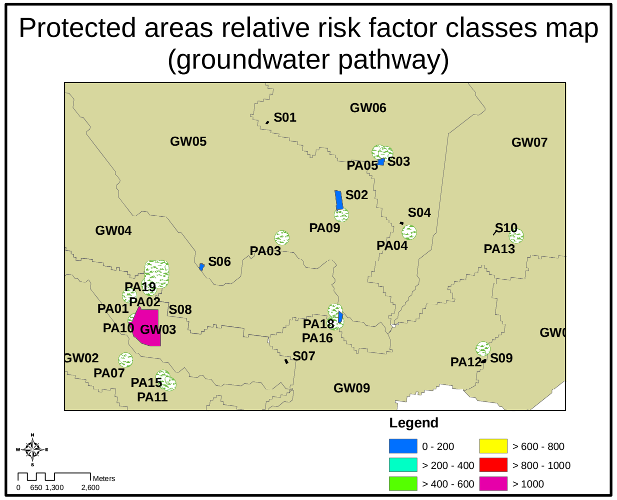


Figure 7.8– Results of the risk factor calculation for protected areas impacted by the contaminated groundwater applied to the Upper Silesia region in the form of relative risk factor classes map. Scores closer or higher than 1000 identify sites of higher risk factor. The black sites are those which do not reach any protected areas.

Figures 7.3 to 7.8 report the results of the risk factor estimation for each potentially contaminated site, in consideration of the spatial relationships between potentially contaminated sites and the identified regional receptors. For all the analysed receptors, S08 results to be the first ranked potentially contaminated sites in terms of relative risk factor. Its relative risk factor scores ranges from the yellow class in the case of surface water receptor, to magenta class in the case of protected area receptors impacted through the wind pathway, the groundwater pathway and the surface water pathway. The high relative risk factor scores for S08 are mainly influenced by the high hazard score (1361,1) and by the fact that S08 impacts regional receptors (H3, GW03 and SW16) with medium -high vulnerability scores, and can come in contact with different PAs (13 on the whole). The other potentially contaminated sites fall in the lower risk factor class, with the exception of S05 and S06 which fall into the cyan class. For S05, 3 protected areas (PA16, PA17 and PA18) are impacted through the wind pathway, the surface water pathway and the groundwater pathway. However, in the first two cases, the pathway scores are high (1 for the wind pathway and 0,94 for surface water pathway (sw14) while for the groundwater pathway the pathway score is lower (0,55 for gw04). In the case of S06 the highest risk factor is estimated for the protected areas reached by the surface water pathway: in fact, 4 protected areas are potentially impacted (PA03, PA16, PA17 and PA18) by way of the integration of sw10, sw13 and sw14 river elements, and the selection of the most conservative value between the estimated surface water pathway scores (i.e. 1).

In the regional relative risk estimation, Equation 5.4 was applied in order to provide two different results. The first result concerns the ranking of the analysed sites through the integration of the relative risk factors for the direct contact pathways (Human Health, Ground water and Surface water) while the second result concerns the ranking of the analysed sites through the integration of the relative risk factors for all the analysed pathways (Human Health, Ground water, Surface water, Protected areas reached by the wind pathway, Protected Areas reached by the surface water pathway and Protected Areas reached by the groundwater pathway).

The direct contact pathways relative risk factors results are reported in Table 7.8 (column 16th) and Figure 7.9, while the regional risk factor results are reported in Table 7.8 (column 17th) and Figure 7.10.

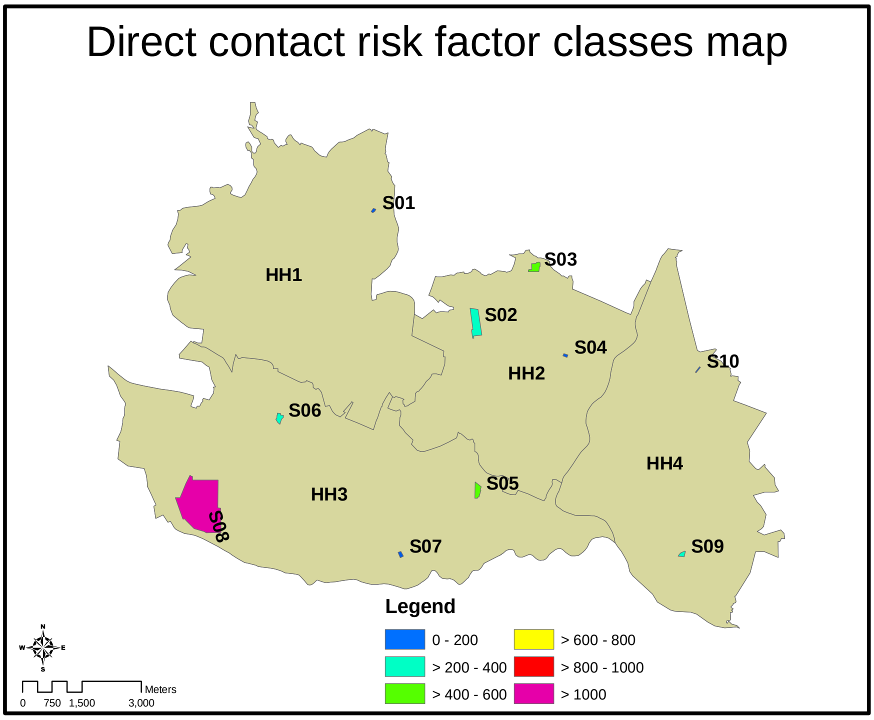


Figure 7.9. Results of the direct contact risk factor calculation applied to the Upper Silesia region. Scores closer or higher than 1000 identify sites of higher risk factor.

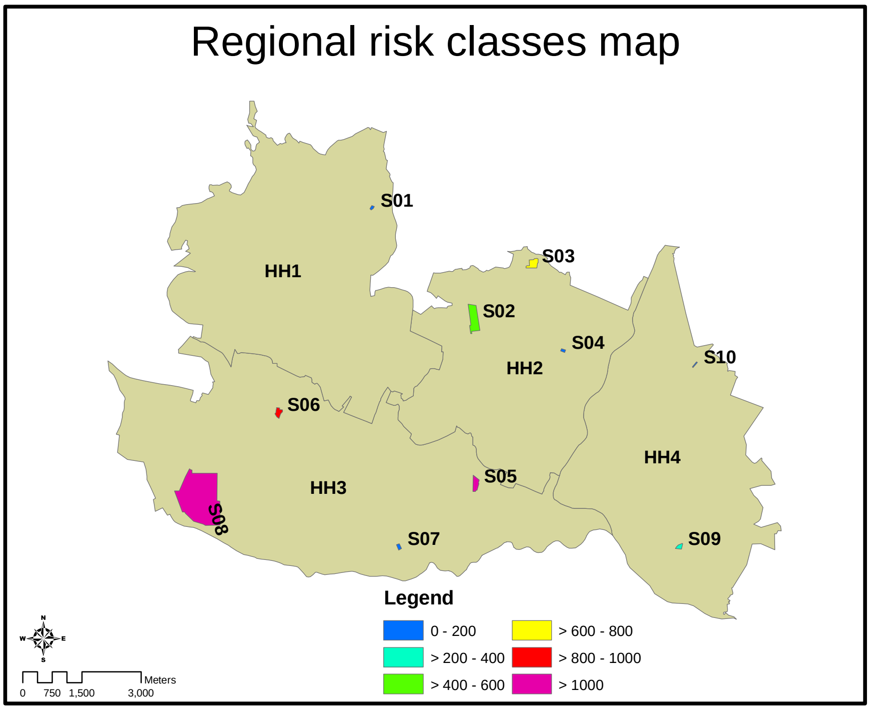


Figure 7.10. Results of the regional relative risk calculation applied to the Upper Silesia region. Scores closer or higher than 1000 identify sites of higher risk factor.

According to the direct contact risk factor estimations (Fig 7.9 and Tab 7.8), the most risky potentially contaminated site is S08, which falls in the magenta class (risk factor higher than 1000) and is followed by S03 and S05 which fall in the green class (risk factor between 400 and 600). The high direct contact risk factor score for S08 is influenced by the very high hazard score (1361,1) and by the fact that S08 can reach H3, GW3 and SW16 which are characterised by medium high vulnerability scores. S03 and S05 gained the 4th and the 3rd position in the hazard ranking, and they reached also human health, groundwater and surface water receptors characterised by medium-high vulnerability, with the exception of S05 which reached GW04 with a low vulnerability score. This influenced also the direct contact final ranking where S03 gained a higher position than S05 though it has a lower hazard score. Moreover, in comparison with S02 which was in the second hazard ranking position, S03 and S05 reached all the direct contact receptors (Human Health, Surface water and Groundwater) while S02 did not impact any surface water receptor, thus reaching a lower direct contact ranking position.

As far as the regional risk is concerned (Fig 7.10), S08 is still the most risky site followed by S05 and S06 which gained a very high regional risk factor since they impact all the identified regional receptors through all the identified pathways. These sites are followed by S03, which falls in an intermediate class (yellow class, risk factor between 600 and 800). Thus the sites which gained a higher regional risk ranking position are those which are characterized by high direct contact score (S08, S03, S05, S06) and at the same time can reach a high number of Protected areas through all the analyzed pathways.

Finally, for each NUTS, a comprehensive regional risk factor was calculated summing up all regional risk factors estimated for all the potentially contaminated sites located in the analyzed NUTS, as reported in Figure 7.11.

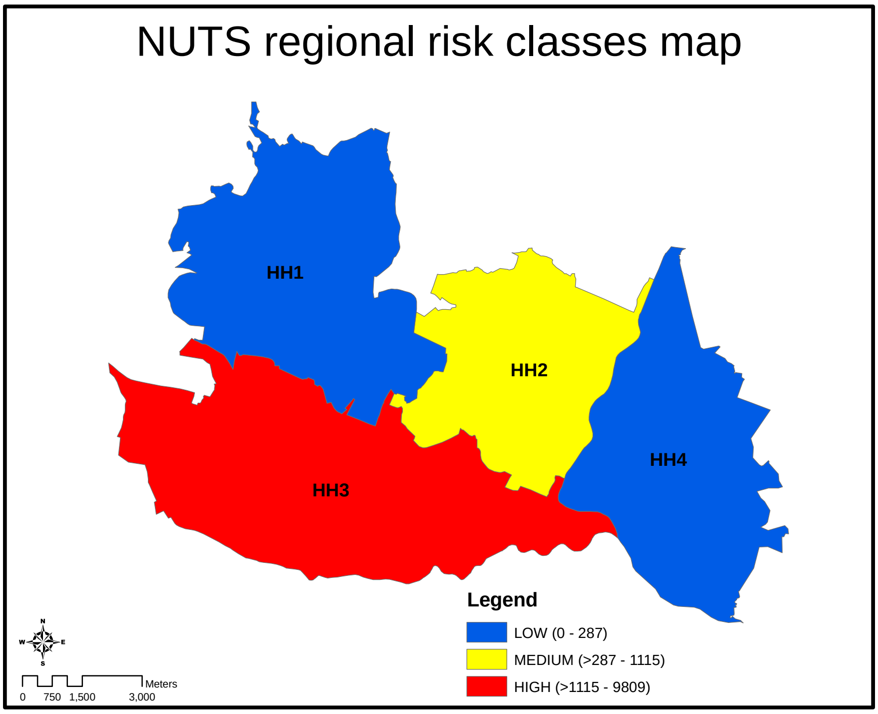


Figure 7.11 – Results of the NUTS regional relative risk calculation applied to the Upper Silesia region in the form of relative risk classes map. Scores are divided in three classes (low, medium, high) defined by the 33 (287), 66 (1115) and 100 (9809) percentiles of the NUTS regional risk scores.

The analysis of Figure 7.11 shows that NUTS HH3 is the most risky administrative unit (NUTS regional risk score higher than 1115) since it contains a high number of potentially contaminated sites (four), most of which gained high regional risk factor scores (S08, S06, S05). NUTS HH3 is followed by NUTS HH2, which contains 3 potentially contaminated sites and falls in the intermediate class (yellow class with regional risk factor score between 287 and 1115). HH1 and HH4 result in the lower regional risk class characterized by very low regional risk scores (lower than 287).

CHAPTER 8

Conclusions

This thesis was triggered by the emerging recognition of the relevance of spatial aspects in the development of sound exposure and risk assessment methodologies for contaminated land management which need to be integrated within GIS-based support systems.

According to the state of the art in risk assessment, risk assessment has a dual role in the management of contaminated: in the planning of the preliminary site investigation phase where a relative risk assessment is required in order to prioritize the potentially contaminates sites to be characterized first and in the full site investigation phase where a site specific risk assessment is performed in order to assess the risks to human health and the environment and to define the site-specific clean-up levels.

Both the two approaches have to increasingly take into account spatial aspects. In literature, on the basis of the scale of the problem to be assessed, two different risk assessment approaches can be identified which deal with the spatial dimension of the problem: the regional risk assessment and the site-specific spatial risk assessment.

The present Ph.D thesis presents the work done to fill the lack of methodologies and tools for the spatial assessment and management of contaminated sites both at regional and at site-specific scale. As far as the regional scale is concerned, a regional risk assessment methodology for the ranking of potentially contaminated sites at regional scale which was developed, applied to a selected case study and implemented within the SYRIADE SDSS (Spatial Decision Support System). At site-specific scale a deterministic and probabilistic spatial risk assessment methodology included within the DESYRE's risk assessment module was implemented and applied to the Porto Marghera case-study.

Going into details, the developed regional risk assessment methodology was implemented in a system called SYRIADE, Spatial decision support sYstem for Regional rIsk Assessment of DEgraded land, which was developed by Consorzio Venezia Ricerche in collaboration with the EC Joint Research Center (JRC).

The proposed direct contact risk factor calculation and the regional relative risk calculation strongly support the national and regional authorities in the ranking of potentially contaminated sites for priority of investigation, when no information on characterization and risk by site-specific methodologies is available.

The regional risk assessment methodology and the resulting SDSS are flexible tools which can be easily adapted to different regional contexts, allowing the user to introduce the regional relevant parameters identified on the basis of user expertise and regional data availability. In fact, the application to the Polish case-study was instrumental in order to test the difficulty of data availability for this type of analysis, but also to prove the adaptability of the methodology in using the data actually at disposal of the users (e.g. the use of the area of the Polish potentially contaminated sites as the size representative parameter for the estimation of the hazard).

Moreover, the MCDA embedded in the methodology supported the effective integration of the expert judgment. In fact, experts can express their preferences and expertise filling in few questionnaires which enable the evaluation and integration of the identified risk components relevant parameters, thus allowing the estimation of the receptors vulnerability and pathway relevance scores. One of the most original aspects of the proposed risk analysis, and also the most useful for decision support, is the spatial feature, which is critical for regional assessments and which was here effectively resolved within the GIS environment. GIS functionalities allowed to spatially link the three essential elements of the risk analysis (source, pathway, receptor) and most of all, to consider all the possible combinations of these three elements within the considered space.

In particular, the vulnerability methodology application to the selected case study proved to be reliable and consistent with the environmental experts expected results. The regional targets and their relevant attributes were identified and spatially characterized. The application of MCDA methodologies allows to take into account both relations between criteria and expert’s judgments and the Choquet integral allows to parameterize non monotonic measures defined by the experts involved in the project**.** Thus, also the developed vulnerability assessment results to be a valuable tool for decision makers facing regional issues.

Nevertheless, some improvements may be identified as well. Among them, a sensitivity analysis would certainly allow to evaluate the influence of the different factors in the overall risk assessment result. Moreover, a methodology for a differentiated weighting of the different factors that compose the regional risk score is another interesting evolution of this procedure.

As far as the site-specific risk assessment is concerned, with reference to the five challenges identified for the site-specific spatial risk assessment presented in Chapter 4, the developed methodology proposed the following solutions oriented to support the formulation of remediation plans for contaminated land. In the hazard assessment phase, the developed methodology supports the selection of CoC with consideration of both their average concentration and peak concentrations, i.e. hot spots. In the exposure assessment phase, it applies geostatistic interpolation methods for mapping the distribution of contaminants concentration which are used in the risk characterization phase in order to provide a zoning of the site based on the risk posed by multiple substances. Each identified risk-based area can be interrogated in order to provide information about most relevant CoCs and exposure pathways. The developed site-specific risk assessment methodology also support the uncertainty analysis, through the application of the Monte Carlo probabilistic calculation of the risk and the generation of maps representing the uncertainty associated to Risk Factor estimates. The final goal of the developed methodology is to support the formulation of remediation plans according to a stepwise spatial allocation of remediation interventions and an on-time simulation of risk reduction performances.

Moreover, the implementation of this methodology was fully supported by an easy-to-use software developed in the popular ArcGIS ESRI geographical information system platform. This risk based GIS software is a part of the original Decision Support System (DSS) named DESYRE. The resulting SDSS was designed to support the risk based remediation of megasites.

It has to be underlined that the current version of the DESYRE software does not allow to include the spatial variability of input values, such as hydrogeological parameters or land use related exposure parameters that heavily affect the risk estimation. However, the same methodological framework can be combined with GIS tools capable to include maps of most sensitive input parameters in the calculation of the spatial distribution of the Maximum Acceptable Concentrations over the site.

As a final remark it should be underlined that even though decision support methods and tools are widely described in literature, their practical use for supporting contaminated land management is not evident (Carlon et al. 2007). This can be the consequence of the site specific complexity of contamination problems and the absence of reliable data which lead to assessment results which are affected by high uncertainty, often not adequately estimated and communicated.

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Estratto per riassunto della tesi di dottorato

L'estratto (max. 1000 battute) deve essere redatto sia in lingua italiana che in lingua inglese e nella lingua straniera eventualmente indicata dal Collegio dei docenti.

L'estratto va firmato e rilegato come ultimo foglio della tesi.

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Ciclo: 21°

Titolo della tesi[1]: SPATIAL AND REGIONAL RISK ASSESSMENT IN DECISION SUPPORT SYSTEMS FOR ENVIRONMENTAL RISK MANAGEMENT

Abstract:

Tradizionalmente nella valutazione dei rischi per l'uomo e per l'ambiente, le relazioni spaziali tra le componenti dell'analisi di rischio e la distribuzione spaziale delle variabili coinvolte non vengono adeguatamente considerate, sebbene esse influiscono sulla valutazione dell'esposizione e quindi del rischio. In base alla scala di analisi, si possono identificare due approcci di analisi di rischio (AR): l'AR spaziale sito-specifica e l'AR regionale. Nella presente tesi di dottorato è stata sviluppata una procedura di AR spaziale sito-specifica che utilizza metodi di interpolazione spaziale per ottenere delle mappe di distribuzione della contaminazione al fine di supportare la zonizzazione del sito sulla base dei livelli di rischio. A scala regionale è stata sviluppata una metodologia innovativa che integra un approccio di AR relativo con analisi spaziali, per selezionare i siti dove le attività di caratterizzazione sono urgentemente richieste. Le due metodologie sono state implementate rispettivamente in DESYRE (DEcision Support sYstem for the REqualification of contaminated sites) e in SYRIADE (Spatial decision support sYstem for Regional rIsk Assessment of DEgraded land) e applicate al sito di Porto Marghera e alla regione dell'Upper Silesia.

Firma dello studente

[1] Il titolo deve essere quello definitivo, uguale a quello che risulta stampato sulla copertina dell'elaborato consegnato.

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Titolo della tesi[1]: SPATIAL AND REGIONAL RISK ASSESSMENT IN DECISION SUPPORT SYSTEMS FOR ENVIRONMENTAL RISK MANAGEMENT

Abstract:

Environmental risks are traditionally assessed and presented in non spatial ways although the spatial relations between the risk assessment components and the spatial distribution of the risk assessment variables strongly influence exposure estimations and hence risks. According to the scale of the problem, two different spatial risk assessments approaches can be identified: site-specific spatial risk assessment and regional risk assessment. In the present Ph.D. thesis the first approach applies geostatistic interpolation methods for mapping the distribution of contaminants concentration in order to support the risk-based zoning of the site. It was implemented in DESYRE (DEcision Support sYstem for the REqualification of contaminated sites) and applied to the Porto Marghera case study. At regional scale, an innovative methodology integrating a relative risk approach and spatial analysis was developed to select sites at regional scale where a preliminary soil investigation is required first. It was implemented in SYRIADE (Spatial decision support sYstem for Regional rIsk Assessment of DEgraded land) and applied to the Upper Silesia case-study.

Firma dello studente

Lisa Pizzol

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