ER - EXTERNAL REPORT

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Environmental Valuation

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1. Introduction

"Environmental valuation involves placing monetary values on environmental goods and services, and on changes in environmental quality resulting from certain actions or inaction" (OECD, 2001, p.24). The main difficulty of and at the same time reason for environmental valuation is that many natural environmental goods and services are not subject to market transactions and their value is not revealed directly by market prices.

Quite a few different methods have been developed to tackle this difficulty. Neither is completely satisfying though. Some of the difficulties are due to the same reasons that it is problematic to do an assessment for environmental damage causing health effects on humans; the "lack of scientific knowledge about causal relationships and lack of relevant data to establish one" (OECD, 2001, p.25). Additional problems arise relating to the assignment of 'value' to the object of potential protection, be it 'non-human biota' or the 'environment' in general. Obviously the first set of difficulties doesn't help the second.

This project has the general objective to 'assess' the environmental exposure and associated risk to non-human biota from liquid discharges from the Belgian Nuclear Power Plants (NPP). This deliverable addresses the question to attribute a monetary value to the radiological impact on the environment and to reflect upon the impact on the non-human biota as compared to the impact on mankind.
At large this leads to three challenging research questions:

1. what ecosystem services[1] are provided in the potentially affected areas and how could they be damaged by the NPP liquid discharges:
    - realm of 'fact':
        - empirical / modelling approach

[1] We follow the Millennium Ecosystem Assessment and Costanza et al. in including both natural and human-modified ecosystems as sources of ecosystem services, and in using the term 'services' to encompass both the tangible and the intangible benefits humans obtain from ecosystems, which are sometimes separated into 'goods' and 'services' respectively (MA, 2003 p. 55-56, Costanza et al., 1997, p.253 - 254).

2. what are these ecosystem services and the potential damage ‘worth’:
    - realm of ‘value’:
        - economic, political, social and normative approach
3. how do we gather and evaluate the information for both former parts, also in comparison to impacts on humans:
    - realm of ‘epistemology’
        - methodological approach

The first research question is limitedly addressed in Deliverables 2 and 3 (Vandenhove et al., 2009), generally treated in chapter 3, and more specifically dealt with for the Scheldt estuary in chapter 5 of this deliverable. The core of this deliverable focuses on the second question. The last question is addressed focussing on the ERICA framework[2], and this implicitly throughout all deliverables, and more explicitly in chapters 2, 3.2 and 6 of this report.

In the first chapter of this report we highlight the role for and the limitations of environmental valuation, as a complement to environmental risk assessment, in helping to make the decision making process inclusive and transparent. We then summarise the scope of environmental valuation and the related approaches. A next chapter discusses applicability issues within the scope of the current research and illustrates with a particular example: the Scheldt estuary. The remaining part of the report offers a reflexion on the radiological protection of non-human biota as compared to the radiological protection of humans.

[2] Other frameworks / approaches are those by IAEA (1992), UNSCEAR (2008), ICRP (2008), etc.

2. *Reasons for Environmental Risk Assessment and Environmental Valuation*

2.1 Environmental valuation as a complement to environmental risk assessment

Environmental Risk Assessment (ERA) involves the examination of risks resulting from practices, processes, products, agents and events that *may* pose threats to ecosystems, organisms and people. Whereas classical risk assessment tends to focus on actual risks, contemporary risk assessment also investigates potential risks, thus recognizing uncertainty as an important concept and taking a proactive stance. The rise of ERA is congruent with a move away from needing to demonstrate harm to the need to demonstrate no harm, and with a general trend within our information society to collect, provision and supply information in order to make decision making as comprehensive as possible, and to increase the accountability of decision makers and industry.

Government, finance, academia and NGOs as well as industry use ERA. Industry uses ERA to implement legislation, but also beyond, to make decisions on materials, processes and facility siting. ERA can be advantageous for several purposes, like the avoidance of liability, the identification of trade-offs when examining comparative risks and the disclosure of financial costs associated with risks (Five Winds International & Pollution Probe, s.d., p.4). In this context, economic Environmental Valuation (EV) of environmental damage is becoming an important complement of ERA. There is a narrow, financial explanation for this, but the rise of the discipline also stems from a more general need within environmental protection policy to set priorities.

Although it is generally acknowledged that broad stakeholder involvement is advisable, if not necessary, for the a priori problem definition and a posteriori discussion of results of concern[3], ERA presents itself as "predominantly a scientific activity and involves a critical review of available data for the purpose of identifying

[3] Also within the ERICA approach used in this report, "the involvement of stakeholders is considered to be good practice and should be encouraged" (ERICA, 2007, chapter 3).

and possibly quantifying the risks associated with a potential threat" (Five Winds International & Pollution Probe, s.d., p.1). As such, ERA in principle aims to be as 'neutral' as possible. However, during all stages of risk assessment, the decision to conduct one in the first place, the drawing up of the framework and methodology, and the evaluation of results, selections are made on both cognitive (e.g. threshold values for measurable effects), normative (e.g. how much remaining uncertainty is acceptable) and possibly emotional (e.g. related to risk perception) aspects. The term 'assessment' includes both evaluation and appraisal, and is thus closely related to 'valuation'.

The activity of ERA as such does not always account for these underlying motives, and may not translate into decision rules on e.g. 'how safe is safe enough' very easily. The ERICA approach (ERICA, 2007), which has been described in Deliverable 1 (Vandenhove and Sweeck, 2009) and used for the ERA as detailed in Deliverables 2 and 3 (Vandenhove et al., 2009a,b) is a tiered approach. Under the first 2 tiers it uses a screening value to screen out sites of no concern (cf. also section 3.2). Although this screening value is helpful in deciding whether further work is required or not, this part of the approach and tool gives little advice on what an assessor or decision maker should do if the screening value is exceeded. Procedures establishing such 'no effect levels' in principle do not aim to convey information on the socio-political and -economic tradeoffs to be made following certain (industrial) activities. They don't give a normative, context bound account on what is good or bad, (un)desirable or (un)justifiable, because there is no common standard to evaluate these categories against. Where the Risk Quotient (RQ) is estimated to be greater than 1 in the Tier 2 level assessment, ERICA does take the ERA further to a Tier 3 assessment. Situations which give rise to a Tier 3 assessment, are likely to be complex and unique. Tier 3 is a probabilistic risk assessment in which uncertainties within the results may be determined using sensitivity analysis. The assessor can also access up-to-date scientific literature on the biological effects of exposure to ionising radiation in a number of different species. Together, these allow the user to estimate the probability (or incidence) and magnitude (or severity) of the environmental effects likely to occur and, by discussion and agreement with stakeholders, to determine the acceptability of the risk to non-human species.

It is also in this broader context that EV of environmental damage is becoming an important add-on to ERA. General radiation protection states that decision making on doses to the environment should be subject to optimisation, based on ALARA 'as low as reasonably achievable', *economic and social factors being taken into account* and provided that the *costs* from further reductions of emissions would *not* be *disproportionate* compared to the resulting benefits. EV can help to substantiate this guideline and its concepts of 'optimisation', 'reasonability' and 'proportionality', by trying to translate the various valuations mentioned (economic, social, environmental…) into a single and thus comparable scale, for which economy offers a monetary one.

2.2 Valuation

'Valuation' and 'value' are conceptually subjective. What's valuable for one person may be worthless to another. Time, place, (cultural) context and personal preferences play a decisive role in valuation, and environmental valuation is no exception in this regard. Fundamental reasoning on why nature is valuable runs between two approaches, an anthropocentric, utilitarian approach, and a biocentric, intrinsic value approach.

With the transition from nomadic to sedentary lifestyles, domestication and domination became central attitudes, and the valuation of nature was mainly determined by its utility for mankind. This approach can be classified under the umbrella of *anthropocentrism*: "elements of nature are valuable insofar as they serve human beings in one way or another" (Goulder & Kennedy, 1997, p.24). The most common theory under anthropocentrism is *utilitarianism,* with the most common description formulated by Jeremy Bentham: valuable is what creates 'the greatest good for the greatest number' (the felicific, utility or hedonistic calculus, focussing on the general concept of 'human wellbeing'). The most commonly used method for environmental valuation, Cost Benefit Analysis (CBA), builds upon this utilitarian principle.

In the 19th century under impulse of Romanticism the valuation of the environment became influenced by *eco- or biocentrism*. This theory rejects the 'dominant species' argument from anthropocentrism (value being dependent on human use), and replaces utility with *intrinsic value*: "value in and for itself, irrespective of its utility for someone else" (MA, 2003, p.6). Put differently, according to this vision all natural things have "intrinsic rights to exist and prosper, independent of whether human beings derive satisfactions from them" (Goulder and Kennedy, 1997, p.26).

Normative discussions about the 'right' basis of valuation are countless and unsettled. We don't need to take a stance let alone solve the issue, but the reason we elaborate a bit on it here, is because *why* we value things influences *what* we value in them and what we don't, and also *how* we do our valuation. It is important to be conscious about this selection chain, both when assessing environmental risks and when valuating environmental damage.

The ERICA approach (ERICA, 2007) which has been described in Deliverable 1 (Vandenhove and Sweeck, 2009) and used for the ERA as detailed in Deliverables 2 and 3 (Vandenhove et al., 2009a,b) implicitly takes up the issue of intrinsic value of ecosystems. Following the questioning of the earlier ICRP paradigm 'if man is protected than the environment is protected', approaches and tools were developed to assess the impact of radiation on the environment, among others the ERICA integrated approach and tool. ERICA does not entail an explicit normative valuation. Exposure situations of concern are merely highlighted. However, the development of an integrated approach and tool to assess the impact on the environment, merely for the sake of the environment, does seem to assign an intrinsic value to the environment (the ecosystems, the specific organisms being its constituent).

Because alternatives to utilitarianism usually only speak about value in theory, but not about practical valuation methods that are applicable to cases in reality, our approach with regard to EV will be to take a very broadly conceived utilitarian approach. We will elaborate on this later (chapter 3). For now, nature can generally be described as the life support system of the economy, and thus of mankind: ecosystems essentially maintain life on Earth (Peirce et al., 2006, p.170). To neglect this role could cause irreversible harm (Costanza, 1991, p.8-9). To assure sustainable policy, it is necessary to include ecological factors to ensure sufficient 'natural capital' for both current and future generations. It is important to substantiate this principle idea, especially when environmental values seem to contradict others, like economic development or energy provision.

This potential contradiction goes back to the roots of the necessity of environmental protection: the ‘tragedy of the commons’ or the discrepancy between our legal system of private property rights and the natural system of common goods. If profit is the leading subjective value driving private property owners, this may cause damaging effects to common goods that have no owner looking after them. Recent evolutions in population growth and ever more impinging technologies only spurred the need to address this issue (growing demands and effects on diminishing resources). All environmental policy thus deals with streamlining subjective values and aims to ensure all possible values (including those we may be unaware of) are taken into account in order to assure balanced trade offs when necessary. It comes as no surprise that economy, the discipline pre-eminently occupied with providing things of an objective value, became a companion of environmental policy. The main difficulty of and at the same time reason for EV is the economic translation of the tragedy of the commons, namely that many natural environmental goods and services are not subject to market transactions and their value is not revealed directly by market prices. A large part of the contributions of ecosystem services to human wellbeing is of purely public goods nature, accruing to people directly, without passing through the money economy. “In many cases people are not even aware of them” (e.g. clean air and water, climate regulation…) (Costanza, d’Arge, de Groot, Farber, Grasso, Hannon, Limburg, Naeem, O’Neill, Paruelo, Raskin, Sutton, van den Belt, 1997, p.257). EV tries to bring to the foreground and translate subjective preferences into an objective and comparable (monetary) measure, and the concept of externalities (consequences that are not part of the decision-making calculus (MA, 2003, p.16) aims to deal with incorporating all possible effects of actions.

2.3 Summary

EV can be described as a tool to make decision making inclusive (internalize all values) and transparent (objectify values by means of a common scale). It can thus aim to substantiate both environmental as well as corporate policies on grounds of intrinsic as well as utility values.

It is important to note that “The mere act of quantifying the value of ecosystem services cannot by itself change the incentives affecting their use or misuse. […] The goal is to improve decision-making processes and tools and to provide feedback regarding the kinds of information that can have the most influence" (MA, 2003, p.20), not to offer monetary solutions to environmental problems. Moreover, results of environmental cost studies reported in a highly aggregated form may encourage the (mis)use of results without full understanding of the assumptions and values that underlie them (U.S. Congress, 1994, p.7). Sufficient emphasis should therefore be placed on reporting the results of earlier phases of the analysis (e.g. emissions and impacts assessments), and on clearly explaining the underlying assumptions.

3. The scope of Environmental Valuation

3.1 EV and its object in theory

As pointed out in the previous chapter, EV can be seen as a complement to ERA and environmental policy in general to aim for inclusiveness on the one hand (accounting for externalities) and to try and translate diverging and/or subjective values on the other hand.

Although the motivation of EV is rather straightforward, immediately several problems arise when trying to perform this sort of valuation in practice. From the outset, it is not possible to begin to estimate costs until it is clear what sort of damage is to be considered or what is the object of protection. Even if the question is restricted to trying to identify the costs of radiological impact to non-human biota in monetary terms, identifying costs of these impacts is not only a matter of selecting and quantifying radiological impacts, but also of assigning values (including, or, in lack of market prices, even consisting mainly of perceived values). Both outstanding issues combine to make EV a very complex and difficult task.

In general, economic assessments tend to "disfavour the 'natural' state in comparison with economically desired alternative uses" for the same ecosystem, "simply because the benefits from the alternative uses are usually more easily measured than are the benefits of ecosystem services. Many of the most important beneficial services of ecosystems are "public goods whose values are not expressed in market prices" (Goulder & Kennedy, 1997, p.42). Commercial or marketed services of ecosystems are "those services that have associated with already established markets in which formal exchange takes place using the medium of money" (OECD, 2001, p. 170-171). Put very generally, pricing of goods and services within existing markets is done on the basis of offer and demand. If the offer is difficult to pinpoint or unknown all together, the demand is extremely varied or even unconscious, and externalities exist, as is the case for many environmental goods, valuation will be anything but an easy task.

A first step within economic valuation of environmental damage is thus to concretize the *offer*. This would mean to map as many different (potential) values or services from a given ecosystem as possible. This framing should ideally provide a clear view on the flow of services depending on the type of ecosystem considered, its condition, on how it is managed and on the socio-economic context. For our case ecosystem services have not specifically been considered in the environmental risk assessment (Vandenhove et al., 2009 a, b). Although site specific reference organisms can be included for the ERA if deemed important and relevant, the ERICA approach works with standardized ecosystems and default reference organisms that largely (but not completely) represent the food-web. As such the ecosystem is represented in a stylized manner, where just a few ecosystem services, like e.g. primary production, are represented by a specific reference organism (e.g. phytoplankton). We discuss this further in section 3.2.

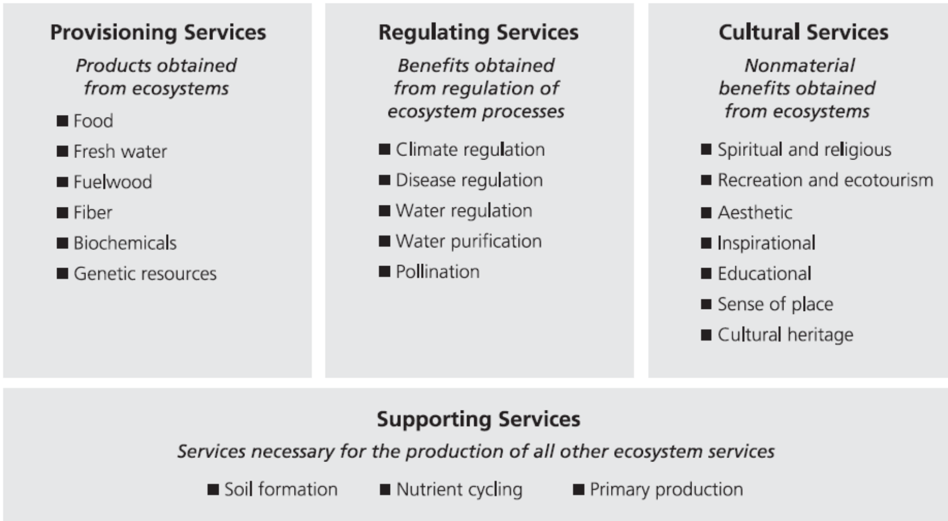
The second task will be to formally concretise the *demand*: how do and can we make use of these environmental services and what do we want or need them for. In other words, how do environmental services of a given ecosystem influence human wellbeing (Costanza et al., 1997).

The most general approach of EV would come down to the following steps. From the combination of listing services and describing their use, a monetary value can be potentially calculated (value A) (how to do this will be dealt with in the next chapter). The next step of EV of damage would then consist of analysing the loss in services and change in utility due to the activity from which damage originates (e.g. through quantifying relevant bio-physical relations), and subtracting the subsequently calculated value (B) from the originally calculated one (A). This monetary translated loss (C) can then e.g. be compared to the cost of mitigation needed to get B closer to A so that C becomes smaller.

Let’s start by investigating what ought to be analysed under A and B.

Commonly three or four broad **categories of ‘ecosystem services’** (including both tangible and intangible goods) are identified (MA, 2003, chapter 2):

Figure 1: Ecosystem Services



Source: MA, 2003, p.57.

A service, not separately taken up in the table but often explicitly mentioned, is the **habitat function**, that describes the role of habitats as nurseries and refugia for populations (e.g. Meire, Herman & Santbergen, 1998, p.315, Costanza et al., 1997, p.254).

For these services, two subdivided **types of utility** are distinguished (OECD, 2006, p.86 – 87 & MA, 2003, p.132 – 133 & Goulder and Kennedy, 1997 & OECD, 2001, p.24 (own composition)):

- *use value* (actual, planned or possible (option value[4]) use)
  - direct use (e.g. eating duck, burning timber… (consumptive use / entailing resource extraction) but also e.g. bird watching, forest hiking (nonconsumptive use / not entailing resource extraction)
  - indirect use (no direct or indirect physical involvement, i.e. “more complex webs of causation” (MA, 2003 p. 73) e.g. plankton)

[4] “to maintain some good in existence in order to preserve the *option* of using it in the future” (OECD, 2006, p.86) This category in fact reflects known and hypothetical future utility of any type of use.

- *non- or passive use value* (existence (no *use* for oneself or for anyone else), altruistic (use for others in the current generation), bequest (future generation) e.g. the existence of the Grand Canyon)

Putting services and use together, roughly most provisioning and cultural services are directly used, and most regulating and supporting services are indirectly used.
Non-use values are the hardest to estimate. The overview nevertheless shows that a very broad utilitarian approach will actually help to substantiate intrinsic values rather than discount them.

To already make a quick application for our case, the use value of Scheldt and Meuse for Electrabel (GDF SUEZ) is obvious, among others through the provisioning service of its water as cooling water (making the energy production process possible) and the regulating service of the rivers serving as waste sink boxes (a recipient of liquid discharges).

To further facilitate economic valuation, these services and uses can be (indirectly) translated into concrete **human benefits**, the individual consumer, from a cost-benefit perspective, being the final actor from who's preferences monetary values can be derived. These benefits to human wellbeing include not being hungry or thirsty, not being sick, having shelter / a place to live, not being afraid, being able to enjoy life, being able to chose how to live ones life, not having to fight with ones neighbours, etc.

The last step is the **monetary valuation of environmental damage** and this would include the revaluation of all previously described steps, but incorporating the alterations due to the influence of certain practices, processes, products, agents and events.

3.2 EV and its object in practice

Formally the previously described steps may seem logical and feasible, but a closer look reveals numerous obstacles and complexities.

"An ecosystem is a dynamic complex of plant, animal, and microorganism communities and the nonliving environment interacting as a functional unit" (MA, 2003, p.49). With regard to the object or offer side of EV, problems thus arise "from the interaction of ecosystem products and services, and from the often extensive uncertainty about how ecosystems function internally, and what they do in terms of life support functions" (OECD, 2001, p.170). When talking about damage, this type of complexity demands a differentiation between intrinsic, stochastic and deterministic factors, and variability, resilience and thresholds are concepts that must be taken into consideration. In general it needs to be acknowledged that ecosystem functioning may be characterized by extensive uncertainty, which compels an attitude of reservation and precaution.

Ecosystems internal complexity gets reflected in its interactions with the economy. The challenge of EV is "to secure some kind of measure of these various ecological-cum-economical values for both natural and semi-natural ecosystems" (OECD, 2001, p.170). Nevertheless the interactive nature of ecosystems is often neglected in applications of EV. Many methodologies apply a bottom up approach, summing up values of individual services and uses. This approach is very unlikely to give an appropriate account of ecosystems complexity. EV of complex ecosystems demands a holistic approach, since "the value of the system as a whole may be more than the value of the sum of its parts. […] A small economic value for one service might suggest it could be despised with, yet its removal could reverberate on the other services through complex changes within the ecosystem" (idem). "Economic value of any one service may depend on its relationship to the other services" (ibidem, p.175).

Valuation will also depend on the spatio-temporal boundaries of ecosystem services, use and damage effects taken into consideration, and whether or not these collide with associated property rights (private goods or local, regional or global public goods (OECD, 2006, p.174).

How does the ERICA approach (ERICA, 2007) which has been described in Deliverable 1 (Vandenhove and Sweeck, 2009) and used for the ERA as detailed in Deliverables 2 and 3 (Vandenhove et al., 2009a,b) deal with the previously described inherent complexity of environmental damage within dynamic ecosystems?

Within ERICA the adverse effects of radiation considered include morbidity, mortality, reproduction, and mutation. Hence both *stochastic* and *deterministic effects* are considered, yet no differentiation between both types of effects is considered in the derivation of the screening value (see further). It should be noticed that many of the endpoints associated with environmental effects are deterministic of nature. EC-PROTECT (Andersson et al., 2008), a follow-up project of ERICA, does only consider effects of radiation on reproduction (the hypothesized most sensitive endpoint) in its derivation of the screening value.

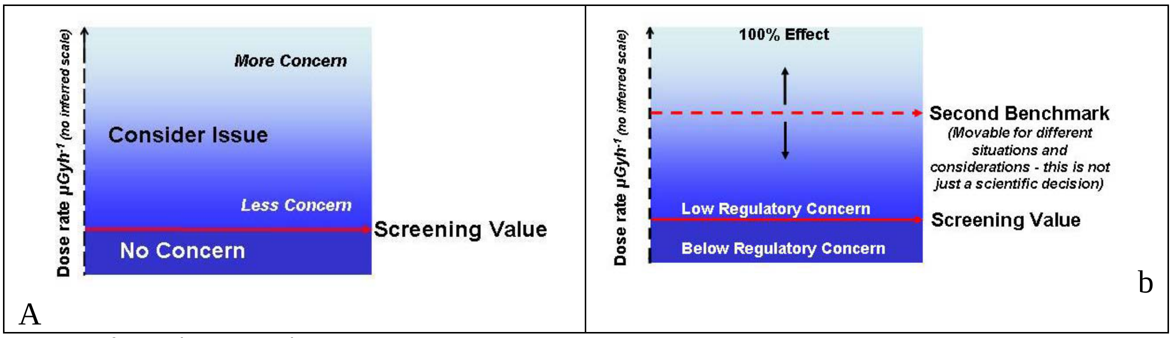
Variability and *uncertainty* are considered within ERICA in a statistical manner. As explained in detail in Deliverable 1 (Vandenhove and Sweeck, 2009), the assessment element of the ERICA Integrated Approach is organised in three separate tiers. Under Tier 1, Environmental Medium Concentration Limits are partially derived using probabilistic calculations (Deliverable 2, chapter 4.2, Vandenhove et al., 2009). The aim of Tier 2 is to identify situations where there is a very low probability that the dose to any selected organism exceeds the adopted screening dose rate. The screening test is thus implemented as follows: (1) An expected value of the RQ is calculated using expected (or best estimate) values for the input data and the parameters; (2) The 95th or 99th percentile of the RQ are estimated by multiplying the expected value of the RQ with an uncertainty factor of 3 or 5, respectively, (reported as the **conservative RQ** in the ERICA Tool). The uncertainty factor is defined as the ratio between the 95th or 99th percentile and the expected value of the probability distribution of the dose rate (and RQ). Where effects are not shown to be negligible under Tier 1 and Tier 2, Tier 3 is a probabilistic risk assessment to be conducted by the assessor who will have to estimate the probability (or incidence) and magnitude (or severity) of the environmental effects likely to occur and to determine the acceptability of the risk to non-human species. It is clear that throughout the latter step, items of complexity and uncertainty that are neglected throughout Tier 1 and Tier 2, like food chain causality, site specific river characterizations (e.g. sediment behavior, the influence of salinity…) etc. will come up as challenging items of major importance. It also should be noted that given the variation between species in ecosystems, it is generally considered not possible (ERICA, 2007, p.19) to develop species-specific assessment systems, as is done for human radiation protection. Within the ERICA Integrated Approach generalised ecosystems and ecosystem representations are used, termed ‘reference organisms’. These “provide a basis for assessing the likelihood and degree of radiation effects” (idem).

Threshold values will also be an important marker, following the non-linear character of ecosystem interaction, especially in the light of irreversible damage. In section 2.1 we already discussed the fact that the threshold values ERICA and its follow-up EC-project PROTECT (Andersson et al., 2008) use, do not provide action levels, but are screening values. The screening value proposed by ERICA and PROTECT is supposed to screen out sites of no concern (Figure 2a). A screening value of 10 µGy h^{-1} incremental dose rate to be applied for all ecosystems and all organisms is suggested by ERICA. This screening value was derived from a Species Sensitivity Distribution (SSD) analysis based on chronic exposure data. The generic screening value was derived using the 5^{th} percentile (or HDR_5) of the distribution of the EDR_{10} (i.e. the Effect Dose Rate giving rise to a 10% effect in the exposed group in comparison to the control group) values to which an assessment factor of 5 was applied to maintain a high degree of conservatism as required for the screening assessments. In other words, the screening value corresponds to a hazardous dose rate affecting five percent of species to a 10 percent effect, to which a safety factor of five has been applied. For the estimation of the generic screening value, data for all general organism- and ecosystem types (terrestrial and aquatic) were used within an SSD. Therefore, the screening value is supposed to protect all ecosystems.

This approach followed by ERICA to derive a screening value is similar to what is applicable for chemicals (European Chemicals Bureau, 2003). EC-PROTECT also proposed a screening value of 10 µGy h^{-1} (Andersson et al., 2008). This value was derived similarly as by ERICA but only including data on reproduction, the most sensitive endpoint (ERICA considered mortality, morbidity, reproduction and mutation as effects).

Using a screening value is helpful in deciding whether further work is required (or not). However, it does not yield information on the character of damage if the screening value is exceeded. A possible refinement is a second, higher, benchmark which identifies, for example, when the risk of impact is ‘significant’ or ‘severe’. This could aid decision making by highlighting where, on the scale of no effect to significant effect, the calculated dose rate is (Figure 2b).

Figure 2: Screening values are used to screen out sites of no concern (a); A second higher could help assessors place their results into context if dose rates were estimated to exceed the screening level. However, the selection of the numeric value of a second benchmark needs to take account of wider societal, economic and political judgments and may vary between situations (b).



Source: after Andersson et al., 2008.

Also, a statistical extrapolation approach could be used to help in the management decision making process by generating different levels of potential impact (e.g. by taking the 50th percentile of EDR_{10} distribution or 20th percentile of the EDR_{25} distribution etc.) (Andersson et al., 2008b). Yet still this doesn't concretise what happens above the screening value, i.e. which damage would occur.

As a consequence, ecosystem dose-response curves linking ecosystem damage with dose rates, which is a requirement for a comprehensive environmental valuation, are not available for our case study.

Resilience is not included within ERICA. However, the derivation of a screening value which is meant to protect ecosystems covers the item implicitly. It is assumed that affecting 5% of the species of a given ecosystem to a 10 % effect level, will not affect the general ecosystem functioning as a whole. It could be questioned if potentially affecting 5 % of the species at a 10 % effect level is justifiable on the long term, as well as e.g. formally allowing 95% of species with 9,5% effect.

3.3 Summary

In summary EV encounters a number of difficulties with regard to its object and scope:

- impracticable to list up all interrelated ecosystem services
    - o very difficult to list up all possible utility (direct (consumptive and non consumptive), indirect, optional use, existence value)
        - ▪ unfeasible to determine every link with every possible benefit
            - • very complex to describe (global) damage

In the following chapter we will investigate how different methods of EV try to deal with these difficulties.

4. Main approaches to Environmental Valuation

In the previous chapter we discussed the complexity of clearly defining the object of environmental valuation and the scope of damage. We have also spoken about the non-marketed character of many environmental services and about the conceptual subjectivity and the variety of underlying reasons for valuation. If both the object in itself as well as the valuation of the object are complex, how then can we remain to sensibly speak about EV?

4.1 Physical linkage methods

One way to try and determine value is to build upon a **physical linkage approach**. Within this approach, EV is truly seen as an add-on to ERA. Physical linkage techniques depend on ERA for an ecological 'dose-response' function. They then turn this function into an economic 'damage function' by technically relating pollution exposure to economically quantifiable parameters like health and productivity changes or restoration costs. One technique for human impact is e.g. the Cost-of-Illness approach, which uses cost of health care, work days lost due to illness, value of human life (morbidity) etc. as quantifiers.[5]

A physical linkage approach thus requires three steps (OECD, 2001, p. 25):

1. Risk assessment to determine a relation between indicators of environmental quality and human health, or productivity changes in bio resources, or material damage
2. This risk assessment leads to dose-response coefficients on the basis of which physical damage can be calculated
3. Following the calculation of physical damage a monetary value can be placed upon the changes in health and/or productivity using prevailing market prices for medical treatments, statistical estimates of value of life, material damage…

With regard to radiological risks, the limitations of ERICA, as described under 3.2, influence phase 1 and 2, and this influence weights heavily upon physical linkage

[5] Other techniques are the Human-Capital approach, the Cost-of-Productivity-Loss approach, and the Replacement-Cost approach, all with their own relevant indicators.

techniques when connecting these phases to phase 3 of monetary valuation. Screening levels do not offer a basis for an EV of damage.

Environmental assessment approaches and tools in general deal with similar restrictions however. Estimating the change in the physical flow of benefits (quantifying biophysical relations) and tracing through and quantifying a chain of causality between these changes in ecosystem condition and human utility, is subject to tremendous uncertainties. “A common problem in valuation is that information is only available on some of the links in the chain, and often in incompatible units” (MA, 2003, p.127). Also, costs associated with changes in the ecosystem, for instance connected to a change in the biodiversity level, may take some time to become apparent or may be apparent only at some geographical distance from where the change occurred (Ketunnen and ten Brink, 2006).

Even if there would not be a lack of scientific knowledge about causal relationships and a lack of relevant data to establish one, physical linkage approaches focus on ‘damage’ related to health and direct production, and, although important, these are but two services ecosystems provide. For many of the other services and uses we have discussed, no market prices are available for a monetary valuation of the physical changes (Schoer, 2007, p.4). For instance, the importance of biodiversity “as a part of ecosystem processes in producing regulating, cultural and supporting services” is not captured in financial markets (Kettunen and ten Brink, 2006). When ecosystem services are owned and traded in the market, users can compare them with substitutes and other commodities, and thus a demand curve can be drawn up based on direct observed market behaviour. Value is thus visible directly through market prices that will reflect a decrease or change in the offer because of environmental damage. Depletion (quantitative (e.g. oil) is easier to valuate than degradation in this regard. In general, the more discrete ecosystem services are and the less direct our use of them is, the more difficult valuation becomes. The main drawback for phase 3 is thus the lack of existing market prices or easily deducible prices.

Another example of a physical linkage approach that particularly tries to avoid the latter problem, is the ***restoration cost approach***.[6] This approach looks at the

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[6] The ***maintenance cost approach*** is similar to the damage cost approach. A maintenance cost approach estimates the hypothetical cost for avoiding pressures on the environment based on existing standards (Schoer, 2007, p.3).

cost of “replacing or restoring a damaged asset to its original state and uses this cost as a measure of the benefit of restoration”. The ‘original state’ is based on “an ecological or expert standard”. Information of replacement costs can be obtained from direct observation of actual costs of restoring damaged assets or from expert estimates of the costs to restore the asset (NEEDS, 2006 p.10-11). Yet this approach requires to take into account causal complexity and spatio-temporal restraints with regard to damage. Moreover it assumes that the ecosystem can be “fully restored back to its original state”, which isn’t always true and doesn’t take into account e.g. existence value (idem).

A methodological approach combining the restoration cost method with the endpoint-oriented life cycle assessment method Eco-indicator 99, is introduced and exemplified in the NEEDS project report (NEEDS, 2006) for land-use changes and chemical atmospheric releases. In these two case-studies, the environmental damage is evaluated on the basis of the PDF indicator (Potentially Disappearing Fraction of species) or the PAF (Potentially Affected Fraction of species), which is subsequently converted to PDF, while the monetary valuation is based on restoration costs. For instance, marginal costs of 0.49 €/(PDF·m^2) were calculated for Germany from the habitat restoration with the least cost resulting in a biodiversity change of at least 20%. Applying such an approach in this project, where we deal with small routine releases, resides in at least the following questions:

→ What is the time frame and which are the two (before and after potential damage occurred) ecosystems under discussion? The ecosystem as is was before any liquid discharges from the NPP’s were released, or the current ecosystem, and the one estimated in 10 years from now, or perhaps at the end of the NPP life-time?

→ Do we know the effects of radiological impact from NPP’s liquid discharges and how can we isolate these effects from other potentially damaging impacts?

→ How can restoration itself be described, i.e. what are the measures/actions needed to restore the ecosystem? Given the natural yearly variability, is introduction of affected species necessary? Is e.g. the (costly) removal of contamination from river sediments justified?

4.2. Behavioural linkage methods

For cases where a physical linkage approach is not applicable (when causal relations aren't that clear and there are no established market prices), **behavioural linkage techniques** have been developed, which are based on individual preferences rather than technical linkages. Behavioural linkage methods rely on the assumption that individuals consciously or unconsciously interpret causal relations between activities, ecosystem services and personal use, and, directly or indirectly express these interpretation through revealed or stated behaviour. EV is thus considered as an independent technique, not as an add-on to ERA. The most common approach to measure individual preferences is to ask or observe the Willingness To Pay (WTP) for environmental improvements and/or the Willingness To Accept Compensation (WTAC) for adverse changes in environmental quality. WTP and WTAC therefore are the pragmatic translation of demand, and these concepts can be used for CBA, "the prevailing approach to ascertaining value" (Goulder & Kennedy, 1997, p.29).

Applied to our case, the valuation would reflect WTP for pollution prevention or reduction (e.g. by restricting releases of radionuclides into the environment) instead of 'actual' direct economic damage from pollution (assessing the radiological impact of released radionuclides).

These methods thus partly solve, or at least deal with the need for alternative valuation when market prices are lacking, but the difficulties with regard to 'what' is to be valuated remain. It is clear from the outset that it is hard to give a genuine valuation about something that is characterized by a high level of assumption, abstraction, complexity and uncertainty. Nevertheless quite a few techniques have been developed for EV within the behavioural linkage approach.

4.2.1 Revealed preference techniques

Revealed preference techniques build upon the observation that *use values* of environmental services (cf. previous chapter) leave a 'behavioural trail' (an observable change in price or quantity due to behavioural change (OECD, 2006, p.87) and can thus be estimated using techniques based on surrogate or existing market behaviour.

Examples of revealed preference techniques are hedonic pricing and travel cost method. Within the *hedonic pricing method* (HPM) the value of a non-market environmental service (e.g. clean air) is estimated by observing behaviour in the market for a related good (e.g. value of property in a given area), namely a market via which the non-market service is implicitly traded. The main applications are non market goods related to property values and to labour markets. For example, an increase in noise will show up in reductions in the value of properties affected by the changes. HPM thus unbundles determinants of a market good, to statistically identify WTP for each constituting characteristic. Accuracy of this method is dependent on the quality and inclusiveness of the information consumers base their behaviour on, and hindered by the multicollinearity of ecosystem damage effects, that make 'teasing out' of independent effects on behaviour difficult (NEEDS, 2006, p.9; OECD, 2006, p.93 – 94). This makes the applicability of HPM for our case, i.e. to value the effects of the isolated aspect of radiological impact from liquid discharges from the NPP's (isolated from e.g. the presence of the NPP in general) questionable.

The *travel cost method* is particularly used for recreational uses of ecosystem services. The method is based on the fact that access to some intangible ecosystem services requires the purchase of market goods. "Specifically, people have to spend time and money travelling to recreational sites, and these costs reveal something of the value of the recreational experience to those people incurring them" (OECD, 2006, p.102). The travel cost method may be a difficult approach to a monetary valuation of environmental damage due to a specific cause, like in our case, liquid discharges from NPP's, to a specific ecosystem service, "since only whole packages of recreational values can be valued by travel expenditures" (NEEDS, 2006, p.10). The holistic methodology of the approach makes it difficult to isolate causes and effects.

4.2.2 Stated preference techniques

Whereas use values can be estimated using revealed preference techniques because they leave a behavioural trail, *non-use values* "can be estimated using the direct method of stated preference techniques, i.e. techniques that are based on questionnaires" that elicit respondents WTP or WTAC (OECD, 2006, p.86).

An example of a stated preference technique is the *contingent valuation method* (CVM), which tries to elicit the WTP (e.g. to maintain an area in its present condition or to improve it) or WTAC by direct questionnaires. This method can reveal both use and non-use values. If people were able to clearly understand the change in environmental quality being offered, and answered truthfully, this direct approach would be ideal. The main critique to questionnaires however, is that of biases (NEEDS, 2006, 2006, p.6; Abaza and Rietbergen-McCracken, 1998, p.5) due to the hypothetical nature of these methods, both with regard to the information offered and the answers given.

4.3 Summary

The inherent subjectivity of valuation remains an issue throughout all EV methods. The utilitarian approach which behavioural linkage methods that use WTP/WTAC are based on, runs into problems more directly in this regard because prices aren't 'out there' yet as they are with physical linkage techniques that use market values. Classical utilitarianism indeed builds upon individual preferences, what something is worth 'for me'. Strong utilitarianism tries to avoid arbitrariness by taking in externalities (the difference between private and social costs), but "society is simply the sum of individuals" (OECD, 2006, p.16) (aggregation). Each person's valuation receives the same weight, no differentiation is made on basis of awareness, education, 'enlightenment'… (Goulder & Kennedy, 1997, p.25, see also 3.1, the part on 'human benefits') which in turn can be accused of another sort of arbitrariness. Also, the utilitarian approach underlying WTP techniques always classifies individuals as consumers, which yields a way of reasoning that is likely to be different from an approach that ask for e.g. the opinion of 'citizens' with regard to 'public goods' (Spash, 2001). On the other hand, unlike physical linkage techniques, non-use values are less prone to fall by the wayside with behavioural linkage techniques (although WTP/WTAC for them will obviously be more hypothetical and context bound).

Physical linkage methods are less vulnerable to a critique of arbitrariness and biases of qualitative methods, but we tried to explain that these methods also rely on selections and qualitative trade offs to investigate damage (e.g. the choice of indicator species, timescale, whether or not to work with substitutability…) and the last step of valuation cannot be conducted without existing markets, which in fact also can be described as the result of WTP/WTAC (De Nocker, Lievens & Broekx, 2004, p.23).

5. Application prospection

5.1 From environmental risk assessment to economic environmental valuation: what is still needed?

We have discussed the restrictions of different valuation techniques related to our case in the previous sections (4.1 and 4.2). Summarized, for the present study, the object of valuation and the potential damage isn’t clear enough to attribute an adequate and comprehensive monetary value to the concrete radiological impact on the environment. Firstly, locating the isolated radiological impact of liquid discharges within a complex web of causal relationships (following internal ecosystem dynamics as well as diverse external impacts) is extremely difficult. Then, to recapitulate, in order to perform a *physical linkage approach*, dose response curves that could allow to draw up a site- and organism specific damage description would be needed, information that the ERICA tool does not provide for our study. Moreover, market prices for many of the ecosystem services such a damage description would refer to, are not (readily) available. Although a *behavioural linkage approach* indirectly addresses the former issue, the isolated and rather abstract issue of ‘potential radiological impact of liquid discharges’ is unlikely to reveal or even incite directly relevant, performative personal preferences and/or behaviour. This makes the potential and moreover the pertinence of WTP / WTAC as a means to EV questionable for the specific scope of our study.

What we will try to do in the following sections is nevertheless to give an idea of the order of magnitude of the monetary value of the natural environment that *could* be influenced by the liquid discharges of NPP’s. This can at least give an indicator of the range of cost of *potential* damage. Costanza and colleagues (1997) have tried to make a consistent and worldwide estimation of the yearly economic value of 17 ecosystem services for 16 different ecosystems, including those relevant for our case, namely wetlands, estuaries and lakes/rivers. They based their calculations on a combination of a variety of the approaches we described in chapter 4, e.g. replacement cost, various WTP approaches, and an extensive literature study of market values.

With the help of this research and the knowledge built up throughout the previous chapters, we will thus give an onset to describe some of the ecosystem services and give a prospective valuation thereof, for the region potentially influenced by the liquid discharges of the NPP in Doel, namely the Western Scheldt estuary and the Verdronken Land van Saeftinghe nature reserve wetland, located a few kilometres downstream the Doel NPP discharge point.

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5.2 The Western Scheldt estuary and the nature reserve of the Verdronken Land van Saeftinghe: Ecosystem description

An estuary is a semi-enclosed body of water, with one or more rivers or streams flowing into it and with a free connection to the open sea. Estuaries are thus subject to both marine influences, such as tides, waves, and the influx of saline water; and riverine influences, such as flows of fresh water and sediment. The Scheldt estuary has a length of 200 km and stretches up to Ghent. The Western Scheldt is the seaward marine section of the Scheldt estuary, commonly described as the section from the Belgian-Dutch border up until the North sea (60 km). It has a width of 1350 metres at Prosperpolder, running up to more than 5 kilometres at Vlissingen (http://www.scheldeschorren.be/natuurtalent/p8.html). Including intertidal salt marshes, intertidal mud flats, shallow water, shoals, and channels, a total surface area of 30930 hectares was drawn up for the Western Scheldt in 1990 (Sistermans and Nieuwenhuis, s.d, p.5).

Along its way, on the left bank, the wetland nature reserve of the Verdronken Land van Saeftinghen is located. A wetland is an area of land whose soil is saturated with moisture either permanently or seasonally. The Verdronken Land van Saeftinghen belongs to the largest and most intact examples of Atlantic salt marshes in Europe. The area has a surface of 3484 hectares. Each tide, the brackish water overflows a large part of the area. The flora there is specifically adapted to these conditions and very unique, including approximately 50 wild plant species (http://www.hetzeeuwselandschap.nl/saeftinghe/index.php). The nature reserve has a very important habitat function. Since 1996 it is a special protected area for birds (1979 Directive 79/409/EEC on the conservation of wild birds).

5.3 The Western Scheldt estuary and the nature reserve of the Verdronken Land van Saeftinghe: Valuation of ecosystem services

Estuaries and wetlands are naturally highly productive and dynamic systems, with a unique functional and structural biodiversity (Costanza, 1997b; Meire et al., 2005). The ecosystem services they provide are multiple (Costanza 1997b; Meire, Herman and Santbergen, 1998, after De Groot, 1997). In the following we will describe some of these services in combination with the work of Costanza and colleagues (1997) earlier referred to (p.28).

Provisioning services

Several products can be obtained from estuaries and wetlands. Western Scheldt is e.g. used for the commercial production of shrimp, sole and cockles (Sistermans and Nieuwenhuis, s.d), but also genetic resources. Costanza et al. used both market-values and informal (non-market) economy information to value these services (Costanza, 1997b).

Regulating services

Estuaries and wetlands, among other functions, play a very important role nutrient cycling and water regulation (e.g. by water buffering of flooding areas thus regulating habitat variations). Especially wetlands "can absorb and recycle large amounts of nutrients and other chemical substances without negative side-effects to the overall functioning of the ecosystem" (Costanza, 1997b, p.12). Particularly the waste treatment function has a considerable economic value. Costanza's et al. estimates are mainly based on cost-saving calculations and costs of replacing the natural function by means of artificial waste treatment. One reference was used in which a survey was conducted to determine the willingness-to-pay for the maintenance of this ecosystem service. Disturbance regulation (flood control and storm protection) values are also "based on estimations of prevented damage or the potential, and in some cases actual, costs of replacing this function of the wetland by artificial constructions" (idem).

Cultural services

Estuaries and wetlands provide services for education, tourism and recreation. E.g. the beaches near Vlissingen and Cadzand, in the mouth of the Scheldt estuary, are extensively used in summer (Sistermans and Nieuwenhuis, s.d). In the Verdronken Land van Saeftinghe there are 12000-18000 (guided) visitors yearly, with at least as many refused due to the sensitivity of the area. In addition around 8000 extra visitors come to the information centre and to walk the corduroy road (De Nocker et al., 2004). Costanza et al. found that little research has been done on cultural services of wetlands. The only references used relate to calculations of the influence of the aesthetic value of wetlands on real estate prices (Costanza, 1997b, p.13).

Habitat functions

"The habitat/refugia function of wetlands is important, both with regard to their value as nursery areas for commercially important species (fish and crustaceans) and as resting and feeding areas for many migratory (and sedentary) species" (Costanza, 1997b, p.12). The Verdronken Land van Saeftinghe is an ideal breeding, resting and wintering place for a very high number of birds. It is of special value as the habitat of animals at a critical stage of their biological cycle, like the moulting and pre-migratory fattening of waders (http://www.wetlands.org/reports/infosheet.cfm?siteref=3NL017). It also plays an important role for the fish and crustacean fauna of the Westerschelde, serving e.g. as nursery for postlarval common shrimp.

Costanza and his colleagues based the nursery value of wetlands on market prices, the habitat value for protection of (migratory) species was mainly derived from willingness-to-pay studies.

Table 1 shows a summary of the calculations done in Costanza et al. (1997), of which we selected the biomes of estuaries and wetlands (and lakes/rivers) and their complementary ecosystem services.[7]

Table 1: Summary of average global value of annual ecosystem services (1994$/ha/yr).

Eco service / **Biome**	Gas regulation	Climate regulation	Disturbance regulation	Water regulation	Water supply	Erosion control	Soil formation	Nutrient cycling	Waste treatment	Pollination	Biological control	Habitat / refugia	Food production	Raw materials	Genetic resources	Recreation	Cultural	Total value per ha ($\$ha^{-1}yr^{-1}$)
Estuaries			567					21100			78	131	521	25		381	29	22832
Wetlands	133		4539	15	3800				4177			304	256	106		574	881	14785
Lakes/rivers				5445	2117				665				41			230		8498

Source: based on Costanza et al., 1997, p.256
Shaded cells indicate services that do not occur or are known to be negligible. Open cells indicate lack of available information.

[7] **Additional info by Costanza on Wetland-functions valuation:** "Within one ecosystem (or biome) some functions are not evenly distributed and we have attempted to correct for these spatial restrictions as much as possible: e.g. recreational activities will focus on the most attractive and accessible parts of the ecosystem so values found for the recreational importance of floodplains or mangroves have not been multiplied for the total surface area but only 30 %.
Within the scope of this survey, it was not possible to make an extensive analysis of all the information available on the functions and values of these biomes and also some wetland functions are under-exposed or not included in the table yet, although their ecological and economic importance is considerable, like their influence on local and even global climate, both through their physical influence on temperature and precipitation, and their influence on gas-exchange with the atmosphere.
Also, except for their importance as nursery areas and migration habitat, little information was found on the economic importance of other biological aspects of the functioning of wetland-ecosystems (e.g. biological control and genetic resources). Thus, the totals should be seen as a very conservative estimate of the total economic value of wetland ecosystems." (Costanza, 1997b, p.11)

Applying Costanza's calculations to our case of Western Scheldt and Verdronken Land van Saeftinghe, leads to the following amounts:

- Wetland value Verdronken Land van Saeftinghe: 3484ha x 14785 = 51 510 940 $/yr
- Estuary value Western Scheldt: 30930ha x 22832 = 706 193 760 $/yr

Clearly these numbers are very rough approaches. Also, the information they yield with regard to our case may seem rather limited. For one thing, the fact that some services taken up in the calculation (e.g. water buffering) are rather unlikely to be influenced by the radiological impact of liquid discharges has to be taken into account. Nevertheless we believe the calculation to be illustrative for EV in general. We also attribute it the merit of raising consciousness about the incontestable monetary value of the environment.

Against the background of the previous criticism, a much narrower approach to conduct some EV for our case could be to apply a replacement cost approach instead of a damage cost approach. Really limiting the research to the case of liquid discharges by the NPP, one could for instance calculate the cost of having to set up an alternative for the use of the naturally available Scheldt water, and of the charges related to having to have the liquid discharges removed, processed, stored and/or transported as waste, avoided by having the rivers as waste sink boxes. Indeed, these both are ecosystem services unaccounted for. Obviously critique on the narrowness of this approach is readily formulated. This only proves that our elaboration on valuation in general, that *why* we value things influences *what* we value in them and what we don't, and also *how* we do our valuation (section 2.2), wasn't unnecessary.

6. Radiological impact on non-human biota as compared to human biota

To reflect upon a comparison between the radiological impact on non-human as compared to human biota, let us start with looking into the differences in our knowledge about these impacts.

Quantifying effects of radiation to non-human biota is a difficult challenge, and although in recent years the topic gets renewed attention of the scientific community, it is clear that there remain major challenges. As shown throughout the previous chapters, when evaluating the impact of radiation to non-human biota, uncertainties on at least three different levels interfere, limiting the significance of the current state of the art of our scientific approach:

1. first the uncertainty in our scientific knowledge about impacts of radiation on the non-human biota itself
2. secondly the lack of knowledge about the complex interactions between risk factors and the complex interdependencies in ecosystems, resulting in uncertainty about potentially far-reaching consequences of relatively minor impacts on specific components of such systems.
3. and finally intrinsic difficulties in the methodological and ethical assumptions underlying the frameworks used to make impact evaluations on ecosystems

In the following, the first two sections summarise the relevant information already given throughout the previous chapters on points 1 and 2, with additional information as regards a comparison with the radiation protection practice for humans. The last section deals with intuitions with regard to point 3, that have been touched upon throughout the report but not worked out yet.

6.1 Uncertainty about impacts on non-human biota as compared to human biota

Previous chapters of this report recapitulate our current knowledge about impact of radiation to non-human biota. This knowledge is based on series of impact studies, but these are limited to a set of 'representative' ecosystems and species, and only some rather rough indicators of impact are considered (mortality, morbidity, reproduction, mutation, etc. of a population, all in function of dose or dose rates,…).

In comparison, much more research has been carried out to get a clear scientific understanding of the radiation effects on humans. For deterministic effects, the causality between radiation and health effects is rather simple, and the impact can be determined for each individual. For stochastic effects, the current scientific knowledge leads to a general impact model used in radiation protection, where impact is assessed by calculating a total effective dose. Based on a linear non-threshold (LNT) impact model, assessments are done by calculating the total effective dose for the whole population exposed. This collective dose (expressed in person.sievert) accounts for all radionuclides and all exposure pathways to man. Based on this collective dose, the number of lethal and non-lethal cancer incidences and the expected genetic effects and birth defects in the population can be calculated using the LNT hypothesis on the dose-response relation (e.g. ICRP, 2007; UNSCEAR, 2000). The risk factors commonly used are (ExternE, 1995):

- *Lethal cancers*: 0.05 per person.sievert
- *Non-lethal cancers*: 0.12 per person.sievert (averaged over all possible cancers)
- *Genetic effects*: 0.01 per person.sievert

Thus, as opposed to non-human biota, dose response curves and rather straightforward corresponding damage descriptions have been developed for human individuals. Although, obviously, putting a price tag on the value of human life and health is subject to considerable and complex considerations, this has clearly helped to facilitate the development of a monetary valuation of radiation effects on human health. When describing physical linkage approaches (section 4.1), we have already discussed e.g. the Cost-of-Illness approach. Such an approach shows that monetary valuation of radiation effects is indeed applicable when dose response curves exist and when the corresponding damage description involves concrete ‘marketed goods’ (like cost of health care and work days lost). Behavioural linkage methods are also applicable. WTP/WTAC will remain subjective but at least some biases can be overcome when its object is as concrete or conceivable as ‘human health’ (as opposed to e.g. ‘potential radiological impact of liquid discharges’ on ‘the environment’).

The ExternE project (1995) used both methods to derive a monetary valuation of health effects. The economic cost of a lethal cancer has been estimated within ExternE by means of the value of a prevented fatality (VPF), often called "value of statistical life" (VSL). This represents the “WTP to avoid the risk of an anonymous premature death”. Typical values of the VSL recommended for policy decisions in Europe and North America are in the range of 1 M€ to 5 M€ (ExternE, 2005). The VSL used in ExternE (1995) was ECU 0.25 million, updated in subsequent phases of the project to around €3 million (€3.1 million in 1998, see European Commission, 1999, p.243), as an average over a number of European studies. If e.g. the WTP to reduce the risk of developing a lethal disease with 1/10000 is 100 €, then the cost of a human life is estimated at 1 M€. The economic cost of non-lethal cancers was estimated based on direct costs (medical care) and loss of income. This value was multiplied with a factor 1.5 in order to estimate the willingness to pay to avoid a non-lethal cancer, the final value obtained being 0.45 M€ (ExternE methodology update, see European Commission, 1999, p. 265). The economic cost of a hereditary effect was estimated under the assumption that it is equivalent to a human death since it will either result in immediate death or a severely impaired life. Therefore the latter cost was considered equal to the 'value of a statistical life' (VSL) (Extern E, 1995, p.69).

Although dose response curves and damage descriptions exist, it should be mentioned that, as for non-human biota, uncertainty nevertheless remains. We can get a feeling about the limitation of knowledge, by elaborating on scientific uncertainty in low dose effects on humans. Despite decades of large scale research on these low dose effects, we observe not only an ongoing discussion in the scientific community about the parameter values and coefficients, but even the dose-effect correlation model is still under debate. It is clear that the knowledge we have about low dose effects on other biota is even much smaller, and yet, as this is the only scientific knowledge available, all assessments are built on this foundation.

6.2 Knowledge gaps with regard to complex interactions in the non-human and human environment

As explained earlier in this report, current approaches to ERA do not model interactions between risk factors nor interdependencies in ecosystems. There is no scientific knowledge available to do this in a direct way. The approach adopted to address this challenge is to work with threshold values designed to guarantee a 'no-effect' on the ecosystem (screening values). These 'no effect' threshold values are based on interpretation and extrapolation of the available studies. The underlying hypothesis is that the impact of environmental doses below such 'no effect' thresholds are likely to be so small that further study is not appropriate, and that efforts for further evaluation are better focussed on other situations with higher exposure. Certainly a common sense approach for priority setting, it is important to stress that this approach does not guarantee 'no effects' in ecosystems, as no study of the interactions or interdependencies is done. Implicitly, the idea is that these hard to model (or hard to understand) effects would get attention if indications of such effects would be available in real world ecosystems.

Comparing this approach with the radiation protection practice for humans, we in fact see a similar attitude: the radiation risk is isolated from other risk factors in order to get a clear scientific understanding of the radiation effects. Radiation protection practice for humans does consider any radiation, however small, as a contribution to possible effects, and leads directly to the ‘as low as reasonably achievable’ (ALARA) approach for exposures implying that any action lowering exposure is useful if costs are not unreasonably high. It nevertheless does not account for complex interactions between risk factors. Recent studies however show that combined effects of multiple stressors such as smoking and living in a 'radon-rich' environment cause health effects not predicted by the current models, and in the future we can expect that the current total-effective-dose / LNT-model for human radiation protection will get refined in order to account for specific combination of risks. Similar progress would be more than welcome for non-human radiation protection approaches.

6.3 Reflection on framework assumptions

Discussing the implicit hypotheses of the framework in which to make ecosystem impact assessments for non-human biota and comparing these to those of the framework for impact assessment on humans is multifaceted.

With regard to human radiation protection, a simplified historical view on the justification in radiation protection, shows the following evolution:

> "… it is clear that until approximately 1950 only deterministic effects where taken into account. These effects only occur above a threshold. So was assured that no risk below that threshold could occur. As a consequence, no justification is needed. The justification effort oscillated since then between qualitative and quantitative approaches. The approach evolved from as low as possible to avoid any unnecessary exposure, over a cost-benefit assessment with the focus on radiation exposure as detriment, to a more pragmatic approach using multi-factorial analysis. The justification effort does not only involve radiological aspects and health, but also economic and social factors, aspects of home security and disturbance. These aspects could be involved in the benefits of the case, as well as in the harmful side of the case." (Govaerts, edited by Loos, 2008, p.59)

It can be said that, with regard to radiological protection of the environment, the evolution of thinking about justification as addressed in the previous paragraph, is rather absent. ERA of radiological impact still works with threshold levels of no effect (cf. supra), and further justification therefore isn't an issue. The release of radionuclides into the environment is implicitly 'justified' by the utility of the source of the release (e.g. NPPs) for mankind.

Historically, and still in use in most current radiation protection practice, only human beings are *explicitly* considered for protection from ionising radiation (Oughton, 2003, p.52). In the past it was generally accepted that if humans were adequately protected, then "*other living things are also likely to be sufficiently protected*" (ICRP, 1977) or "*other species are not put at risk*" (ICRP, 1991). Over the last decade the view that non-human biota are protected when human biota are, has been questioned however. This happened in part because of increasing world-wide interest in environmental sustainability, and in part because of recognition of situations where non-human biota may be exposed but exposure to people is limited by restricted access to an area or by other protective measures. Many initiatives were started on the part of the radiation protection authorities to challenge the established approaches that were too exclusively directed at human protection of the individual, to broaden the point of view to include protection of the non-human environment.

So radiological protection of the non-human environment has been put on the agenda, but clear differences in points of departures and approaches remain. This is so mainly on two interconnected key components of justification: intrinsic value and the value of an individual.

For human individuals intrinsic value is in general well established, 'every person counts'. As discussed previously, the radiation protection assessments indeed count for individual health effects, and the radiation protection policies differentiate between doses acquired in voluntary situations (professionally exposed persons, medical exposure) against doses where people have no choices of avoiding it (general population dose limits). This frameworks tallies with the ecocentric perspective of intrinsic values, where all parts of nature have a categorical character for which there are no substitutes, their value is a value in itself, individually unique and thus non replaceable (cf. 2.2).

But for non-human biota the focus lays on the population level and even more, the focus is mostly on the services of these populations for mankind. These points of departure are explicitly part of the impact assessment methodology, as illustrated by some examples :

- It has always been generally accepted that if humans were adequately protected, "*other living things are also likely to be sufficiently protected*" (ICRP, 1977) or "*other species are not put at risk*" (ICRP, 1991). In 1991, ICRP added the following clarification *"Occasionally, individual members of non-human species might be harmed but not to the extent of endangering whole species or creating imbalance between species*" (ICRP, 1991).
- The basic assumption, such as formulated in the ExternE methodology, is that "the total impact [of the exposure] on a population, due to the loss of individuals, can be relatively unimportant if the proportion affected is relatively small or if the region where the impacts occur is relatively small, allowing for migration of healthy individuals into the affected areas" (ExternE, 1995, p.61).

Literature discussing the impact of the contamination in Chernobyl indirectly follows the latter way of reasoning (e.g. Expert Group on Environment (EGE), 2005). It is recognized that both individual and population effects caused by radiation-induced cell death were observed in plants and animals, but stressed that by the next growing season following the accident, population viability already substantially recovered as a result of reproduction and immigration.

To continue, we have already explained how, to reduce the complexity in ecosystem modelling, it is assumed that a limited set of organisms is representative for the whole ecosystem. Moreover, often a difference between functional redundant species and species that make a unique or singular contribution to ecosystem functioning is commonly made within ERA and EV, on the grounds of which substitutability can be taken into account ("impact created by the loss of one or more species is compensated for by others" (MA, 2003 p.61-62), e.g. within the functional group of species for nitrogen fixation).

Summarized protection of individual members of species is not necessarily assured within current frameworks for the assessment of radiation impact on non-human biota, and focus is put on population resilience. While the object of human impact assessment is the individual person, the object of the assessments of non-human biota are 'non-redundant species for an ecosystem'.

6.4 Summary: The weight of radiological impact on non-human biota as compared to human biota

When asked to compare the weight of effects of radiological impact to non-human biota as compared to humans, it is tempting to try and convert both to a monetary value and discuss the obtained results. However, we have shown the inherent difficulty to convert ecological as well as the human health 'value' into monetary terms, and how different methodological points of departure lead to relative incomparability from a theoretical point of view. Moreover, the comparison of the two is open to criticism in itself. Indeed, how can we compare the impact of e.g. "1 in 10,000 excess cancer risk to human health with the unspecified probability that a particular species of beetle will become extinct?" (Peterson, 2005). For both cases reflection on justification is necessary, a reflection that will necessarily involve scientific, socio-economic and ethical, as well as epistemological aspects.

7. Conclusion

Throughout this report on economic Environmental Valuation we have tried to answer the following questions: *why* would one apply an EV, *what* is its object, and *how* can it be conducted. The first question, dealt with in chapter 2, turned out to be the easiest one: valuation lies at the heart of all decision making, and in the light of its inherent diversity and subjectivity, EV can try to help make decision making inclusive (by internalizing as many values as possible) and transparent (by objectifying different values by means of a common (monetary) scale). In fact all the difficulties encountered during the next two chapters can be interpreted as proof for this need for inclusiveness and transparency. In chapter 3 we elaborated on the complexity of ecosystem dynamics and risk factors, and the difficulties and limits Environmental Risk Assessment (in our case conducted with ERICA) encounters to reflect and substantiate this complexity. In chapter 4 we elaborated on different techniques for EV, broadly divided between physical and behavioural linkage approaches, and subsequently discussed the ambiguity of putting a 'correct' monetary value on a hard to pinpoint object.

Summarized, for the present study, the object of valuation and the potential damage isn't clear enough to conduct a proper EV, i.e. to attribute an adequate and comprehensive monetary value to the concrete radiological impact on the environment. Firstly, locating the isolated radiological impact of liquid discharges within a complex web of causal relationships (following internal ecosystem dynamics as well as diverse external impacts) is extremely difficult. Then, in order to perform a *physical linkage approach*, dose response curves that could allow to draw up a site- and organism specific damage description would be needed, information that the ERICA tool does not provide for our study. Moreover, market prices for many of the ecosystem services such a damage description would refer to, are not (readily) available. Although a *behavioural linkage approach* (based on individual preferences rather than on scientific linkages) addresses the former issue, the isolated and rather abstract issue of 'potential radiological impact of liquid discharges' is unlikely to reveal or even incite directly relevant, performative personal preferences and/or behaviour. This makes the potential and moreover the pertinence of WTP / WTAC as a means to EV questionable for the specific scope of our study.

The merits of EV as a complement to ERA and more general an aid to inclusive and transparent decision making are clear: the different issues that caused the inability to conduct one for our case, clearly and purposively indicate the need for further research. In chapter 5 we still provided a case study, not for attributing a monetary value to the radiological impact on the environment, but, more broader, to give an idea of the order of magnitude of the monetary value of the natural environment that *could* be influenced by the liquid discharges of NPP's. We applied the findings of Costanza et. al together with the knowledge gathered throughout the previous chapters of our report to the case of the Scheldt estuary (Western Scheldt and Verdronken Land van Saeftinghe). This led us to an estimated total value of 757 704 700 $/year (1997 figures), serving as an indicator of the range of cost of *potential* damage.

In the last chapter we brought together all the previous items in a critical evaluation of the methodology used for non-human as compared to human biota. All radiological protection is about seeking an 'optimal' risk level. This means one that provides an 'adequate' level of protection. But defining concepts like 'optimal' and 'adequate', necessarily also involves the social, political, ethical and economic context in which decision making takes place to be taken into account. Human radiation protection is starting to acknowledge these various dimensions of valuation. There is an evolution towards to a more pragmatic approach using multi-factorial analysis, that allows aspects other than radiological aspects and health to be included (like the disturbance of the family situation, or the financial situation of the patient). This evolution should also be encouraged within non human radiation protection, consequently forcing to go beyond the level of thresholds of no effect towards an inclusive exercise of justification. The latter should go beyond simply implicitly allowing potential effects to the environment for the sake of human utility of the source of radiation. Once again, the utility of EV is clear, to help ensure inclusiveness and transparency by making sure that as many different criteria and respective values are included in decision making and translated into a common scale for comparison between different policy options. Just as for human radiation protection, we would thus encourage a combination of physical and behavioural linkage methods for non-human biota too, to allow both quantitative and qualitative knowledge to be taken into account.

Further scientific work to refine our knowledge in order to be able to conduct a serious EV on radiological impacts following liquid discharges from NPPs thus is needed, and not only in studying the impacts (the concrete potential damage) in controlled lab experiments and in real life ecosystems, but also in developing a more general critical reflectivity about impact assessment frameworks for environments unrelated to a human perspective. Both will inevitably influence the feasibility and outcomes of EV, but at the same time the ideological rationale of EV can also assist in making the research and its framework more inclusive, transparent and realistic. EV will never be easy, but initiatives such as this one by Electrabel (GDF-SUEZ) are encouraged, because it has become more than obvious that simply stating nature is priceless won't grant the environment the protection and recognition it is worthy of and requires.

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