Law and Economics
of
Environmental Damage Assessment

Een rechtseconomische analyse van milieuschadebeoordelingen

Proefschrift ter verkrijging van de graad van doctor aan de Erasmus Universiteit Rotterdam op gezag van de rector magnificus

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On the cover:

Jan Fabre, 'The Man Who Measures the Clouds' (1998), Bronze, Photograph: Philippe De Jaeger

A metaphor for the artist, who tries to capture the impossible in his work

To my families, my dearest friends and my mentors

Thank you all because you saw in me what I was unable to see in myself

I could not have made it through without all of you

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Foreword

This thesis represents an attempt to establish a dialogue between economics, law and ecology. It contains information on various economic techniques that can be employed to value environmental damage in litigation. It highlights the importance of weighing the advantages to be obtained by economic activities against the damage that they may cause to the environment and biodiversity at any level of law and regardless of political boundaries.

It is hoped that the underlying idea of bridging separate disciplines will inspire specialists belonging to different areas of expertise. The primary objective of this thesis is to provide novel perspectives on a topic that has been the subject of much debate for centuries: the valuation of natural resources. It also seeks to lay the foundations for future cooperation between scientists, economists and lawyers. In order to advance the law on the environmental damage assessment, it is necessary to conduct further research on economic valuation methods, to encourage interdisciplinary collaboration, to provide training for experts and to establish networks to exchange ideas and to takeaways from the practice. This will ensure optimal deterrence of accidents to the environment and cost-effective restoration of degraded ecosystems and ecological functions across the world.

Table of Contents

The polluter pays principle is meaningful
only if one can establish satisfactorily
how much that should be.
(Chapman and Hanemann, 2001)

C’est à la fois l’anarchie et l’arbitraire
qui caractérisent l’évaluation par le juge.
(Memlouk, 2010)

It is not very good,
but it is the best we have.
(Fankhauser and Tol, 1999)

CHAPTER I

Introduction

The PhD Project In A Nutshell

1. The starting point: the first ICJ environmental compensation decision

The initial point of departure for this research is represented by a case initiated before the International Court of Justice (ICJ) in 2010, which resulted in the first decision of the ICJ on the topic of environmental damage and compensation.

On 18 November 2010, the Republic of Costa Rica brought an action against the Republic of Nicaragua, claiming that Nicaragua's army had invaded, occupied and used Costa Rican territory, as well as causing significant harm to Costa Rica's protected rainforests and wetlands. This was as a result of the construction of a canal and certain works of dredging on the San Juan River by Nicaragua.

Subsequently, on 22 December 2011, Nicaragua initiated legal proceedings against Costa Rica for violations of Nicaraguan sovereignty and major environmental damage on its territory, resulting from the road construction works in the border area between the two countries along the San Juan River. In light of the principle of judicial economy, the ICJ joined the proceedings by two separate Orders in 2013. The final judgment on the merits was delivered on 16 December 2015. The Court determined that Costa Rica exercised sovereignty over the disputed territory and that Nicaragua had violated Costa Rican territorial sovereignty by excavating three caños (i.e. channels, galleries) and establishing its military presence on Costa Rican territory. Consequently, Nicaragua was obliged to make reparation for the damage caused by its unlawful activities on Costa Rican territory. The ICJ granted 12 months to the parties to reach an agreement. However, the parties held divergent views on the methodology for the assessment of environmental harm. Costa Rica claimed US$ 6,711,685.26 as compensation for quantifiable environmental damage caused by Nicaragua's excavation of the 2010 caño and the 2013 eastern caño, in addition to the costs and expenses incurred to monitor and remedy the damage. Conversely, Nicaragua requested that Costa Rica be compensated for a sum of no more than US$ 188,504, with the understanding that this figure represented only material damage (i.e., damage to property and other interests of the State that could be quantified in financial terms). Despite the parties' best efforts to reach an agreement on the compensation, they were unable to do so. Consequently, the Court settled the matter on 2 February 2018. The Court awarded US$ 120,000 for the impairment or loss of environmental goods and services in the period prior to recovery (including pre-judgment and post-judgement interests), plus US$ 2,708.39 for the costs of restoration.

What is peculiar about this case is the methodology adopted to assess the environmental harm. Costa Rica proposed an 'ecosystem service approach' based on a report by a Costa Rican non-governmental organisation, Fundacion Neotropica, which maintained that environmental damage might be calculated on the basis of the reduction or loss of the ability of the environment to provide certain goods and services.

Nicaragua, for its part, proposed a less complex method of assessment which involved an ‘ecosystem service replacement cost’. This was in terms of which Costa Rica was only entitled to the cost to replace environmental services that either have been or may be lost prior to the recovery of the impacted area. In the end, the ICJ rejected both methodologies proposed by the parties because they were unreliable and considered that an overall assessment of the environmental damage was more appropriate. The approach permitted the consideration of correlations between more and less severe harm to the same natural resource. Furthermore, it was more suitable for wetlands, which are among the most complex ecosystems. Finally, it could take account of the capacity of the area for natural regeneration. It is also noteworthy that the ICJ decided not to appoint independent experts despite the disagreement on the methodology of environmental damage assessment.

2. The state of the art

The case of Costa Rica v. Nicaragua suggests that there might be uncertainty in the law on how to assess environmental damage, with special regard to the heads of damages, the remedies and the specific methodologies of compensation. However, it is unclear where this uncertainty originates. It appears that the environmental damage assessment is a common topic between at least four different domains of knowledge:

- Tort law and economics,
- Environmental economics,
- Tort law,
- Ecology.

The areas above are interrelated and each contributes to our understanding of environmental harm. A fundamental distinction in tort law and economics is the one between damage and damages. While the term "damage" refers to the harm or loss caused by an accidental event, the term "damages" refers to the amount of monetary compensation due by the liable party. Consequently, the term 'environmental damage' refers to the loss of natural resources caused by an accident, whereas the expression 'environmental damages' refers to the monetary compensation due by the polluter. This dissertation will employ these terms consistently with the scholarship of law and economics.

It is important to note that an important conclusion of this literature is that injurers are expected to behave optimally if the expected liability (damages) equals (or is approximately the same as) the expected harm (damage). Indeed, this would induce them to optimally internalise ex ante the full social costs of their harmful activities. Scholars of law and economics have been studying in depth the possible causes of divergence between damage and damages. One of these is that an accident may cause both pecuniary and non-pecuniary losses, the latter being more difficult to measure. Likewise, environmental accidents may cause both material damage that is monetarily quantifiable and non-material damage that is more difficult to quantify with money.[1]

Moving to the field of environmental economics, the valuation of natural resources has been the subject of increasing investigation, with the objective of informing not only ex ante benefit-cost analyses for policies and projects, but also the ex-post valuation of environmental accidents. A significant challenge in this field since the 1970s has been the assessment of what have been termed ‘non-use’ or ‘passive-use values’ of environmental goods and services. The concept of 'use values' refers to the utility that people gain from the environment for utilising it in a material sense (i.e., extracting and selling materials, purchasing a ticket to visit a park, etc.). Even when people do not utilise the environment directly, they still value the possibility of doing so in the future or they care about the fact that future generations will benefit from the same possibility. The latter values are known in environmental economics as ‘non-use’ or ‘passive-use values’ of environmental goods and services.[2] The issue with these values is that traditional valuation methods are not applicable because they are based on observable behaviours, such as the choices of people that may result in changes to market prices that can be easily measured. Given that non-use values do not often translate into market behaviours, economists developed new techniques to value non-market-based losses. These techniques

[1] See chapter II, §2 and §3, for a comprehensive literature review on this point and more explanations.
[2] See chapter II, §4 for more references on this point.

have been progressively improved to make them more accurate and reliable. However, their reliability is still heavily debated.[3]

In the field of tort law, the environmental damage has traditionally been the object of legal claims related to the recovery of financial losses. However, given the limitations of these tools when it comes to remedy (and prevent) harm to natural resources, environmental liability provisions have been increasingly employed to address the harm caused by oil spills, toxic leakages and other polluting events, together with regulations, administrative and criminal sanctions. The adoption of environmental liability statutes commenced in the United States in the 1970s, underwent a significant acceleration in the 1990s, and entered a period of full-scale implementation in the European Union in the 2000s. A primary innovation was the conferral of legal standing upon public bodies (trustees in the United States and public administrations in the European Union) to file claims for damage to publicly owned natural resources.

Another significant advancement has been the expansion of the scope of compensable environmental damage to encompass explicitly non-use or passive-use values, thereby establishing the obligation of the polluter to provide compensation. This occurred both at the international level and in the United States.[4] However, a recent tendency at the national level is represented by the restoration-based compensation of environmental damage. In such cases, polluters are often obliged to restore the environment instead of paying an amount of money equivalent to the harm caused to the environment. This approach can avoid the contentious and more time-consuming use of methods designed for non-use values. Nevertheless, there might be several limits to this approach.

In addition to the aforementioned developments, ecologists have been engaged in the development of their own valuation methodologies since the 1990s, with a heightened focus during the past two decades.[5] From 2001 to 2005, the Millennium Ecosystem Assessment (MEA) involved more than 1,360 experts from around the world in an endeavour to assess the condition and trends in the global ecosystem. In 2007, the United Nations Environment Programme (UNEP) launched a comprehensive global assessment of ecosystem services, resulting in the 2012 TEEB (The Economics of Ecosystems and Biodiversity) Report. In 2022, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), an intergovernmental organisation established to facilitate dialogue between science

[3] See chapter II, §5 for an in-depth analysis of each category of methods of non-market valuation in environmental economics.
[4] See chapter IV and V for an excursus of these legal developments.
[5] See chapter III for a review of this literature.

and policy, released its assessment report on the multiple values of nature. This report provides guidance on the design and embedding of diverse values of nature in policy-making.[6]

3. The gap and the research question

While the fields of tort law and environmental economics have evolved rapidly and influenced each other,[7] the issue of full compensation for environmental damage remains a less explored topic in both fields. Moreover, scholars of tort law and economics have not yet examined the efficiency of the more recent restoration-based compensation compared to the monetary one in terms of optimal deterrence. In light of the existing academic gap and the urgency presented by the current ecological emergency, this dissertation aims to contribute to the literature by investigating whether current remedies for environmental liability are providing polluters with optimal care incentives to minimise the environmental costs of accidents while, at the same time, ensuring cost-effective restoration.
This general question branches out in the three following sub-questions:

1. Do current methodologies of environmental damage assessment induce optimal deterrence and cost-effective restoration?
2. Are remedies in legislation and case law inducing optimal deterrence and cost-effective restoration?
3. How can remedies for environmental harm be improved to induce more efficient deterrence and cost-effective restoration?

4. The research method

This dissertation employs an interdisciplinary research method that integrates economic analysis of environmental liability law with a comparative approach.

[6] IPBES (2022). Methodological Assessment Report on the Diverse Values and Valuation of Nature of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Balvanera, P., Pascual, U., Christie, M., Baptiste, B., and González-Jiménez, D. (eds.). IPBES secretariat, Bonn, Germany.
[7] It should be noted that the language of environmental liability laws adopted worldwide reflects the evolution of the non-market valuation and the ecosystem services literature at different points in time. This statement comes from C.A. Jones and L. DiPinto, 'The Role of Ecosystem Services in USA Natural Resource Liability Litigation', 29 *Ecosystem Services* 333, at 334 (2018).

A positive economic analysis is conducted to address the first two sub-questions. This is one of the two principal methodologies employed in the field of law and economics, and is a prerequisite for further discussion on the optimal legal framework.[8] Such method is adopted because it allows to predict the consequences of certain methods of environmental damage assessment on the behaviour of polluters and the other parties involved (e.g., insurer, public authorities, etc.). In view of that, liability laws are analysed from the mainstream economic perspective, i.e. rational choice theory.[9]

However, the principal methodology employed in this study is comparative law and economics. This approach integrates economic analysis with comparative law, as both disciplines adopt a non-state-centric perspective on legal analysis.[10] The outcome is noteworthy, particularly given the dearth of economic considerations in contemporary comparative law on both sides of the Atlantic Ocean. Conversely, the comparative approach may be absent in mainstream law and economics. It has been observed that the interest in comparative law is relatively recent among scholars of law and economics. Increasingly, legal scholars attempt to explain legal changes through economic analysis, while economic scholars attempt to explain economic results through comparative legal analysis. Three specific reasons motivate the methodological choice of this dissertation.

Firstly, environmental issues and accidents are naturally and often situated across various levels of law and beyond national jurisdictions.[11] Therefore, environmental law is inherently comparative and multi-level.

Secondly, differences and analogies among legal systems may be often tracked to specific interest groups who take advantage of the law and who influence certain results. By comparing different laws on remedies for environmental damage, both the interest groups behind the law and those receiving incentives by the law become clearer.

Furthermore, the comparative approach is useful in producing policy recommendations based on real examples and in identifying more desirable outcomes in terms of efficiency (i.e., maximisation of social welfare).[12]

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[8] A.M. Pacces and L.T. Visscher, 'Methodology of Law and Economics', in B. van Klink and S. Taekema (eds.), *Interdisciplinary Research into Law,* at 85ss (2011).

[9] See chapter II, §1.

[10] U.A. Mattei, L. Antoniolli & A. Rossato, 'Comparative Law and Economics', in *Elgar Encyclopedia of Comparative Law* (2006).

[11] This is in line with the multi-level theory proposed by S. Cassotta, Environmental Damage and Liability Problems in a Multilevel Context. The Case of the Environmental Liability Directive (2012).

[12] See chapter II, §1, footnote 10 for a discussion on this point and more references.

As a lighthouse, the economic analysis of tort law illuminates the path, transforming a sterile comparison of laws into a profound and expansive comprehension of the actors, their private interests at stake, and the tangible consequences of often intertwined laws.

5. The structure

The structure of this dissertation is tripartite, with each part addressing one of the three sub-questions above (§3). The first part proposes an economic framework to analyse the existing methods of nature valuation in environmental economics and ecology. It draws conclusions on the efficiency of methods of environmental damage assessment from a theoretical perspective. The second part looks at the law at the international and domestic level to determine whether the existing remedies fit in the previous economic framework. Two empirical chapters on oil spills and climate change damages provide further insight into the analysis of the law presented in the text. These chapters highlight additional factors that play a role in determining the most effective remedies. The final part of the dissertation builds on all the previous findings and draws conclusions on optimal remedies that are likely to fully internalise the environmental costs of accidents.

	Title	Research question	Research method	Area
II	Valuing Environmental Damage: Fundamental Issues and Methods	Do current methodologies of environmental damage assessment induce optimal deterrence and cost-effective restoration?	Literature review of environmental economics and tort law and economics	Environmental Economics
III	Restoration versus Damages: an Optimal Order of Remedies			Tort Law and Economics
IV	Testing The Efficiency of Remedies on The Bench of International Courts	Are remedies in legislation and case law inducing optimal deterrence and cost-effective restoration?	Positive economic analysis and comparative law and economics	International Law and case law
V	Comparing The Efficiency of Remedies at The Regional Level: US v. EU			US Law and case law, EU Law and case law
VI	Environmental Damage and Oil Spills			International Law, national environmental laws and case law
VII	The Ecosystem Services Approach			Ecological Economics
VIII	Climate Change ? L'Addition, s'il Vous Plaît ! A Comparative Law and Economics Account on the Calculation of Damages			Climate science, climate economics, climate-tort cases, national laws
IX	Conclusions	How can remedies for environmental harm be improved to induce more efficient deterrence and cost-effective restoration?	Normative economic analysis	Tort Law and Economics

PART I

Do current methodologies of environmental damage assessment induce optimal deterrence and cost-effective restoration?

CHAPTER II

Valuing Environmental Damage: Fundamental Issues and Methods*

One of the most significant challenges facing humanity in the present era is the destruction of the natural environment. When this destruction is caused by human activity and is deemed to be a direct result of risky practices, liability can play a role in addition to all other regulatory and market-based tools. From a legal perspective, liability is primarily aimed at providing compensation to victims, whereas from an economic perspective, its main objective is to act as a deterrent. This can be achieved by inducing potential polluters to invest in care (ex ante) in order to minimise expected losses (ex post). However, several issues may impede the achievement of either compensation or deterrence. This chapter examines how different methods of damage assessment in litigation can enhance or undermine the goal of environmental liability from an economic perspective. The overarching research question is whether the methods of environmental damage assessment that are available for courts and that have been already employed are likely to pursue deterrence in an efficient manner. The chapter is structured as follows. Firstly, the theory of tort law and economics is reviewed to explain how damages should be theoretically assessed in order to achieve deterrence. Secondly, the advantages and drawbacks of traditional methods of environmental damage assessment are illustrated based on the scholarship of environmental economics. Thirdly, conclusions are drawn based on the comparison of these methodologies in order to provide judges with a cost-effective and 'on average' accurate valuation technique. The final section introduces the recent tendency of the law to move towards restoration as the primary remedy, which will be investigated in greater detail in the subsequent chapter.

1. Introduction

'Recent widespread damage of oil spills in Europe suggests that the current legal and economic framework does not provide a mechanism for preventing oil spill damages'.[1]

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* This chapter has been presented in the Online Seminar Using the Law to Save the Planet: Legal Options to Address Climate Change and Ecological Destruction, organized by the Erasmus School of Law and held online on 20 May 2022. It has been later published in the Erasmus Law Review - Special Issue 'Using the Law to Save the Planet' n. 3/2022. A first version of it has been previously presented in the EDLE online internal seminar of 3 June 2020.

[1] R.T. Carson and S.M. Walsh, 'Preventing Damage from Major Oil Spills: Lessons from the Exxon Valdez', 32(3-4) *Oceanis: Serie de Documents Oceanograthiques* 351 (2006).

It hardly needs explanation that environmental accidents lead to huge costs for the society and, thus, they require adequate measures to prevent and compensate them. If we look at the existing tools to tackle the environmental harm from the perspective of an environmental economist, we can see a general distinction between command-and-control regulations and market-based instruments. They all play a role to control environmental pollution when, due to high transaction costs, private parties cannot bargain and address market failures.[2] However, the two classes of instruments largely differ. Command-and-control tools (conventional approach)[3] consist of regulations to force firms and individuals to uptake a share of pollution-control burden irrespective of the costs.[4] They include uniform standards (technology and perform-based standards).[5] On the other hand, market-based instruments aim to induce firms and individuals to undertake pollution control in a more cost-effective way[6] through price signals, such as tradable permits and pollution charges.[7]

[2] W. Pfenningstorf, 'Environment, Damages, and Compensation', 4(2) *Law & Social Inquiry* 347 (1979). When market decisions affect third parties (those who are not involved in that specific market transaction) by causing negative externalities, a market failure occurs. Pollution is a typical example of market failure. Actors causing negative externalities should take into account the full social costs of their production, otherwise they keep engaging in activities leading to pollution levels that would be higher than what is socially optimum. Externalities might be internalised through private negotiation or (oftener) government intervention. Indeed, pollution is considered to be the 'fundamental theoretical argument for government intervention'. *See* R.N. Stavins, 'Environmental Protection and Economic Well-Being: How Does (and How Should) Government Balance These Two Important Values?', in J.A. Riggs (ed.), *How Do Business, Government and Media Balance Economic Growth and a Healthy Environment?* (2003), at 1.

[3] On the reasons why regulatory instruments became so frequently adopted to control environmental pollution, *see* N. Keohane, R. Revesz & R.N. Stavins, 'The Choice of Regulatory Instruments in Environmental Policy', 22 *Harvard Environmental Law Review* 313 (1998).

[4] Stavins (2003), above n. 2, at 4.

[5] Design standards require the use of technologies, while performance standards determine the maximum amount of pollution that firms or individuals are allowed to emit (ibid., at 4).

[6] At least in theory, market-based tools to control pollution are more cost-effective because they induce behavioural changes while minimising the social costs to pursue the predetermined levels of pollution. For an in-depth view of costs of regulation versus liability, *see* S. Shavell, 'Liability for Harm versus Regulation of Safety', 13(2) *The Journal of Legal Studies* 357 (1984).

[7] For an extensive review of environmental market-based instruments, *see* R. Stavins, 'Experience with Market-based Policy Instruments', in K. Mäler and J. Vincent (eds.), *The Handbook of Environmental Economics* (2003), at 355.

Private liability laws belong to this last category since they can provide potential polluters with strong incentives (implicit prices)[8] to consider the consequences of their actions[9] and, thus, to efficiently prevent accidents.[10]

According to the theory of tort law and economics, the primary economic goal of liability laws is therefore to induce polluters to adopt optimal levels of care and activity so that the total social costs of accidents are minimised.[11] In other words, the first aim of liability laws is the optimal deterrence of environmental accidents and not (only) victims' compensation.[12] Scholars of law and economics have been writing for years on how liability laws should be designed to induce optimal deterrence. In this chapter, one of the possible causes of inefficiency is addressed, i.e. the mismatch between (expected) liability and (expected) harm. Although the

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[8] T.S. Ulen, 'Rational Choice Theory in Law and Economics', in B. Boudewijn and G. De Geest (eds.), *Encyclopedia of Law and Economics. Volume I. The History and Methodology of Law and Economics* (2000), at 790ss.

[9] In law and economics, it is traditionally assumed that human beings take 'rational' decisions, meaning that people choose the options that best meet their preferences given certain expectations that they create based on the optimal amount of information that they gathered. In this way, human beings are assumed to maximise their expected utility. This predominant approach to human behaviours is called 'rational choice theory' and it is predominant in law and economics, although heavily debated because of several limitations. *See* H. Schäfer and C. Ott, *The Economic Analysis of Civil Law* (2004), at 53ss. A relatively more recent approach, the so-called 'behavioural law and economics', assumes instead that people do not act always rationally due to psychological biases, such as the 'endowment effect' for which people are willing to pay less for acquiring something (a right or a good) than what they are willing to accept for giving it up. Based on this and more psychological findings, this approach tends to support a more regulatory approach rather than believing in private market transactions. *See*, *ex multis,* C. Jolls, C.R. Sunstein & R. Thaler, 'A Behavioral Approach to Law and Economics', 50 *Stanford Law Review* 1471 (1998).

[10] An economic outcome is regarded as "efficient" if there are no other feasible outcomes under which every party is better off. This is the classic definition provided by Vilfredo Pareto (see Schäfer and Ott, above n. 9, at 23).The Pareto efficiency criterion is generally regarded as a "minimal" normative requirement, as it does not allow the lawmaker to select a unique outcome (the set of the Pareto efficient outcomes tends to be extremely large). An alternative efficiency criterion, developed by Nicholas Kaldor and John Hicks, requires the lawmaker to select the policy outcome that maximises the difference better the benefits and the costs accruing to the parties. This criterion is based on a monetary representation of preferences (a party prefers outcome A because she is willing to pay more to get to A). The Kaldor-Hicks criterion lies at the foundation of cost-benefit analysis (see D. Pearce, G. Atkinson & S. Murato, *Cost-Benefit Analysis and the Environment. Recent Developments*, OECD, 2006). This criterion maximises the net benefits of a policy outcome independently of how benefits and costs are distributed across the parties. For this reason, it has been heavily criticized. Cost benefit analysis is said to be deaf to equity concerns (see for instance ch. 15 of Pearce, Atkinson, Murato). Advocates of cost benefit analysis reply to this criticism by noting that equity concerns can - and should - be addressed by a different policy tool: taxes and subsidies (L. Kaplow & S. Shavell, 'Why the Legal System Is Less Efficient than the Income Tax in Redistributing Income', 23(2) *The Journal of Legal Studies* 667 (1994). Under this approach, lawmaking should maximize the monetary gains for society. These gains become then available for redistribution according to equity criteria. See S. Shavell, *Foundations of Economic Analysis of Law* (2004), at 3-4.

[11] According to Calabresi, the primary function of tort law is to reduce the sum of accident costs and costs to avoid accidents (minimisation of social costs). This reduction goal then applies to three categories of costs. The first category (primary costs) concerns the costs of accidents themselves and the costs to avoid accidents; the second category includes the costs of inefficient distributions of costs within the society and the costs to spread the risk of accidents (distribution). Tertiary costs lastly refer to the cost of administering the treatment of accidents (costs of litigation, for instance). *See* G. Calabresi, *The Costs of Accidents: A Legal and Economic Analysis* (1970), at 26-27.

[12] To understand why deterrence is likely to minimise the costs of accidents and thus maximise social welfare, legal rules need to be regarded as creating implicit prices for alternative behaviours. More specifically, tort damages (or a criminal fine) represent a price for infringing the law. Given that an increase in prices normally produces a decrease in demand, an increase in legal price, e.g. tort damages, should theoretically induce potential polluters to a decrease in unlawful behaviours (Ulen, above n. 8). Knowing that a certain amount of damages has to be paid as a consequence of the accident, potential polluters will be induced to adjust their levels of activity and precaution in such a way that the additional private cost (including the probability of future damages) is lower than the additional benefit. *See* A.M. Pacces and L.T. Visscher, 'Methodology of Law and Economics', in B.M.J. van Klink and S. Taekema (eds.), *Law and Method. Interdisciplinary research into Law (Series Politika, nr 4)* (2011), at 95.

meaning of these terms is readily summarised in the next paragraph, it is sufficient to underline this crucial fact: if the liability falls short of the harm, the incentives to minimise the total costs of accidents are expected to be inadequate. The problem is that environmental accidents pose serious issues of uncertainty about the level of losses and these issues become clear especially when assessing damage in litigation. This is due to a number of reasons that will be illustrated in depth. Although environmental economists developed methods to quantify the harm to nature, they all present pros and cons in terms of accuracy[13] and costs. Possible inaccuracies are likely to undermine the possibility to achieve optimal deterrence of environmental accidents through liability, hence leading to more pollution. The aim of this chapter is therefore to determine whether, from a perspective of law and economics,[14] there exists a methodology of environmental damage assessment that can be regarded as sufficiently accurate but also cost-effective to induce optimal deterrence in environmental liability laws. In order to answer this question, some basic notions of environmental tort law and economics, such as 'accident' and 'expected liability', are first introduced. Then, the theory of tort law and economics is reviewed to clarify how the damage should be assessed to achieve optimal deterrence. Building on this theoretical framework, existing techniques to value natural resources in environmental economics are illustrated, with special regard to their advantages and shortcomings. Thereafter, they are compared in view of pursuing deterrence in an efficient way. In conclusion, despite the inexistence of a general consensus in economics for a fully accurate and cost-effective methodology of environmental damage assessment, it can be argued that there is possibly room for improving the deterrent effect of environmental liability laws by employing more accurate methods in case of large accidents causing a considerable loss of non-use values of nature.

2. Starting from the economic meaning of terms

From an economic perspective, the term 'accident' generally refers to the harmful outcome (i.e. loss of utility) of events that neither the injurer nor the victim wanted to occur, although they might have affected its likelihood and severity.[15] 'Accidents' occur without being

[13] A central goal in the valuation of the environment is to produce accurate value estimates (*see* infra). Since the 'true value' is unobservable, like in many other disciplines, criteria need to be developed as indicators of accuracy. Reliability and validity are the common criteria of accuracy in environmental economics. Reliability has to do with variance and erratic results, whereas validity refers to unbiased results.

[14] For the sake of clarity, this article adopts the mainstream approach to the economic analysis of law, i.e. the 'rational choice theory' (*see* above n. 9). Alternative approaches (e.g. the behavioural one) would deserve separate examination and they are not considered here.

[15] S. Shavell, Economic Analysis of Accident Law (1987), at 1.

intentionally induced and between parties that are not previously bound by a contractual relationship.

Moreover, they hold a peculiar reciprocal nature, meaning that both parties (injurer and victim) are responsible for the resulting harm.[16] This means that both of them can affect accident risks by exercising care and accidents are bilateral in nature. The opposite scenario is when only one party's behaviour (injurer) can affect accident risks by exercising care and accidents are thus unilateral.

In environmental accidents, injurers (e.g. the polluting companies) unintentionally cause harmful effects that could have been reduced by adopting ex ante optimal decisions on the levels of care and activity. However, also victims of environmental accidents may reduce the costs of accidents by adopting care and, for instance, moving to another location in order to be free from negative externalities. For this reason, environmental accidents are often bilateral.

Other important terms to define are those of 'liability' and 'expected liability'.

With 'liability' (or damages) we mean the amount of monetary compensation for which the injurer is legally liable towards the accidents' victims, whereas the 'expected liability' (or expected damages) is the loss multiplied by the probability of suffering that loss.[17]

According to the theory, injurers are expected to behave optimally if the liability (damages) equals (or is approximately the same as) the harm (or damage).[18] If, for instance, there is more than one possible level of harm (stochastic loss) and the liability equals the actual level of harm, also the expected liability will match the expected harm[19] and parties' behaviours will be optimal.[20] The next section will delve more into the economic rationale underlying this theory.

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[16] R. Coase, 'The Problem of Social Cost', 3 *The Journal of Law and Economics* 1, at 13 (1960). For instance, the victims of industrial emissions have the option to move to another location in order to be free from the negative externality, that would determine a certain cost for them. If they move, then the polluter could avoid the cost of precaution to avoid the harm and it would be optimal that the victims move if their cost is lower than the cost of precaution for the polluter.

[17] Shavell (1987), above n. 15, at 6.

[18] Shavell (2004), above n. 10, at 236. However, under the negligence rule the optimal magnitude of damages can be even higher or lower compared to the magnitude of harm, because injurers can avoid liability by taking due care (as long as the due care is set optimally). For this reason, law and economics scholars agree that only under a strict liability regime economic efficiency requires that the injurers pay for all the losses they caused. *See*: R. Cooter 1984, 'Prices and Sanctions', 84(6) *Columbia Law Review* 1523, at 1542 (1984); W.M. Landes and R.A. Posner, *The Economic Structure of Tort Law* (1987), at 64; R.A. Posner, *Economic Analysis of Law* (1986), at 176; A.M. Polinsky, *An Introduction to Law and Economics* (1983). On the other hand, even under negligence a too low level of expected liability might induce injurers to prefer being liable rather than taking due care.

[19] Shavell (2004), above n. 10, at 236. The underlying assumption according to the rational choice theory is that parties have an optimal amount of information about the level of harm and they know in advance if the accident may result in more possible levels of harm.

[20] L. Kaplow and S. Shavell, 'Economic Analysis of Law', in A.J. Auerbach and M. Feldstein (eds.), *Handbook of Public Economics Vol. 3* (2002), at 1661.

3. The economic relevance of accuracy in environmental damage assessment

Incentives to minimise accidents' costs are theoretically optimal only where the expected liability equals or is approximately the same as the expected harm.[21] The economic rationale for the match between expected liability and expected harm is that polluters tend to invest in care up to the point where the marginal cost of risk reduction (or precaution) equals the marginal benefit (avoided loss or expected liability). The logical consequence is that if the liability is lower than the harm (not all social costs are internalised), potential polluters will underinvest in care, which turns out into underdeterrence and higher likelihood of accidents. Conversely, if the liability exceeds the loss resulting from the accident, potential polluters will invest in care more than what is socially desirable, which means for instance a too low level of activity.[22] As a consequence, deviations between the level of (expected) liability and the level of (expected) harm will distort the incentives to minimise the total social costs of accidents.[23] It thus makes sense to understand why deviations would occur and how much accuracy is socially desirable. Possible causes of divergence[24] include information asymmetries between parties about the magnitude of harm, courts' errors, low levels of polluters' assets and difficult-to-estimate components of harm, such as non-pecuniary losses. Non-pecuniary losses are components of losses which have no economic price or value on markets (i.e. health damage).[25] Nevertheless, they are regarded as compensable with money in tort law,[26] hence raising either fundamental (why compensate non-pecuniary losses) or more practical questions (how to value non-pecuniary losses) that have been largely debated in law and economics. Setting aside the 'why' that has been already examined (deterrence purposes)[27] and the 'how'[28] that will be the

[21] This is specifically true under strict liability (*see* above, n. 18) and for unilateral accidents, i.e. when it is assumed that only the injurers' behaviours (and not the victims' ones) can influence accident risks. *See also* Kaplow and Shavell (2002), above n. 20. In bilateral accidents, strict liability alone cannot achieve the socially optimal goal because it does not provide the victims with care incentives and other liability rules are needed. See Shavell (2004), above n. 10, at 182ss.

[22] For more detailed examples, *see* Kaplow and Shavell (2002), above n. 20.

[23] However, negligence rules represent an exception to that, *see* above n. 18.

[24] A. Endres, Environmental Economics: Theory and Policy (2010), at 62ss.

[25] S.D. Lindenbergh and P.P.M. van Kippersluis, 'Non Pecuniary Losses', in M. Faure (ed.), *Tort Law and Economics, Vol. 1, Encyclopedia of Law and Economics* (2009), at 215.

[26] Ibid.*,* at 217 for references to studies that show the importance of nonpecuniary losses in awarding tort damages.

[27] Awarding compensation for nonpecuniary losses is socially desirable to give parties the right behavioural incentives. 'All costs of accidents should be charged to those who could avoid them by taking precautions', *see* M. Adams, 'Warum kein Ersatz von Nichtvermogensschaden?', in C. Ott and H. Schäfer (eds.), *Allokationseffizienz in der Rechtsordnung* (1989), at 213.

[28] For a review of approaches to nonpecuniary losses referred to personal injuries, *see* Lindenbergh and van Kippersluis, above n. 25, at 223ss.

object of the next section, it is now important to understand how much accuracy is socially worthwhile, considering that parties in liability lawsuits hold opposite private interests.[29]

A first largely agreed point in tort law and economics is that there is not one optimal rule for all situations.[30] The efficiency of damage awards necessarily relies on the specific circumstances. Arlen proposes five main criteria[31] to classify and analyse these situations: harm to replaceable versus irreplaceable goods; unilateral versus bilateral risk; strict liability versus negligence; individual versus vicarious liability; lastly, further issues: information costs, uncertainty, judgement proof [32] problems. For instance, a strict liability regime requires that the injurers pay for all the losses they caused, whereas this is not true under negligence.[33] Suffice it to say, the full compensation of losses should not be seen as a goal in itself but as a means to achieve optimal prevention taking into account the specificities of the case at hand.

Another important point emphasised by law and economic scholars is that, as a general rule, liability should not grossly and systematically deviate from accidents' social costs.[34] Slightly inaccurate assessments are acceptable provided that the expected liability is on average correct.

The third point is that accurate assessments of damage levels increase the administrative costs to handle related cases.[35] In order to save costs in litigation, abstract assessments might help and they should be preferred to the extent that they provide a good approximation of real losses and that the saved costs (benefit) outweigh the costs of small mistakes (accidents' costs that go uncaptured).[36] Consistently, difficult-to-estimate components of harm would be correctly replaced by average estimates if the cost of their precise estimation outweighs the benefit of their inclusion (for instance, they are too small compared to the harm).[37] This is also applicable to the non-use values of nature (see next section).[38]

[29] 'The primary objective of the plaintiff is to collect as much as possible and that of the defendant is to pay as little as possible' (L. Kaplow and S. Shavell, 'Accuracy in the Assessment of Damages', 39(1) *Journal of Law and Economics* 191, at 191 (1996)). Also, consider that what will be said is applicable both to accidents resulting in trials and to settlements (*ibid.,* at 198).

[30] J. Arlen, 'Tort Damages', in B. Bouckaert and G. De Geest (eds.), 2 *Encyclopedia of Law & Economics* (2000), at 682.

[31] Ibid.

[32] 'Parties who cause harm to others may sometimes turn out to be judgement proof, that is unable to pay fully the amount for which they have been found legally liable.' From: S. Shavell, 'The Judgement Proof Problem', 6 *International Review of Law and Economics* 45 (1986).

[33] *See* above n. 18.

[34] M.G. Faure and L.T. Visscher, 'The Role of Experts in Assessing Damages – A Law and Economics Account', 2(3) *European Journal of Risk Regulation* 376, at 378 (2011).

[35] Ibid., at. 379.

[36] Ibid.

[37] L.T. Visscher, 'Tort Damages', in M.G. Faure (ed.), *Tort Law and Economics, Vol. 1, Encyclopedia of Law and Economics* (2009), at 160.

[38] S. Shavell, 'Contingent Valuation of the Non-use Value of Natural Resources', in J.A. Hausman (ed.), *Contingent Valuation: A Critical Assessment* (*Contributions to Economic Analysis), Vol. 220* (1993), at 371.

The fourth point is that It is important to take into account the information held by injurers when they decide on precautions.[39] If injurers know exactly the level of harm they will cause when taking decisions on care and activity levels, accuracy in damage assessment influences their behaviours and it makes economic sense for the court to measure harm accurately.[40] Conversely, if injurers lack knowledge in advance (like in many environmental accidents), very accurate assessments in litigation would increase the administrative costs without providing injurers with better incentives (social loss).[41] Lastly, it is also true that accuracy incentivises injurers to learn about the harm before they act, for that they can adopt a level of care in line with the expected harm.[42] Then, ex post accuracy in assessing damage is socially desirable if injurers can anticipate the magnitude of loss ex ante and it is socially optimal for the injurers to get that piece of information.

To conclude and going back to the original question (how much accuracy is socially worthwhile), broadly speaking, injurers should pay for all the harmful effects of their actions (including pecuniary and non-pecuniary losses) under strict liability. Rough estimates have to be preferred if they considerably lower administrative costs and they serve to assess components of loss that are not big enough to overweigh the costs to assess them.[43] Rough estimates should also be preferred if injurers lack ex ante information about the loss. Conversely, more accurate estimates should be preferred if they allow to assess very large components of losses caused by accidents and such losses are big enough to outweigh the costs to assess them. These conclusions carry over to the difficult-to-estimate components of environmental losses that will be analysed in the next section.

4. The challenge of valuing natural resources in economics

Having reviewed the fundamental scholarship of law and economics on the accuracy of damages, the next step would be to understand why issues of inaccuracy may occur when dealing with the environmental damage. It might be helpful to begin from a general

[39] Kaplow and Shavell (1996), above n. 29.
[40] Ibid., at 194 (proposition 1), but this is true 'if it is not too costly for the harm to be observed by Courts'.
[41] In the words of Faure and Visscher: 'A more accurate damage assessment ex post would therefore not necessarily result in better behavioural incentives ex ante' because polluters adapt their behaviours to the 'estimation' of the losses they expect to cause (*see* above n. 34, at 379).
[42] Kaplow and Shavell (1996), above n. 29.
[43] If the law totally excludes these elements from the magnitude of liability, a social loss might occur. Indeed, the injurer will not invest in optimal care to avoid the loss that nobody is legally entitled to claim. As a consequence, part of the magnitude of harm is likely to remain unprevented unless other tools are set down by the legal system to respond to the undeterred negative externality (regulations, criminal fines, taxes, etc.).

understanding of how values are assessed in economics. Value has been the topic of different disciplines: philosophy, anthropology, sociology, psychology and economics. Because of that, it is not surprising that value has many meanings.[44] When it comes to 'environmental' values, philosophers specifically examined the notion of intrinsic values,[45] psychologists developed methods to assess how much people believe in intrinsic values and economists tried to measure economic values that could be used to take decisions on how to manage natural resources.[46] Economics defines the environment as valuable in two senses: in terms of its direct impact on individual utility and in terms of its impact on production.[47] Utility is an economic concept used in neoclassical economics to measure the well-being of people; it refers to happiness or satisfaction of individuals.[48] For instance, people derive utility from buying certain goods and, thus, the value of goods is given by their change in utility (marginal utility). This is also applicable to environmental goods.[49] People can indeed derive utility from carrying out activities in nature, such as birdwatching or swimming. As a consequence, the level of

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[44] T.C. Brown, 'The Concept of Value in Resource Allocation', 60 *Land Economics* 231, at 231 (1984). Brown classified all values into preference-related and non-preference-related (i.e. values in mathematics). Preference-related values include: intrinsic, instrumental, functional, held and assigned values; they all involve a human preference, i.e. 'the setting by an individual of one thing before or above another thing because of a notion of betterness' (*ibid.,* at 234).

[45] In the words of Brown (above n. 44, at 234) intrinsic values are "when the valued entity is an end in itself and its value is independent of any other entity". In the words of M. Lockwood (*see* 'Humans Valuing Nature: Synthesising Insights from Philosophy, Psychology and Economics', 8(3) *Environmental Values* 381, at 384 (1999)): 'it is a widely shared intuition for which an accepted theory to support it is yet to be developed'.

[46] For an overview of the contributions from all these disciplines on human values for natural resources, *see* the seminal work by Lockwood, above n. 45, at 382. He drew on Brown (above n. 44) and J. O'Neill, 'The Varieties of Intrinsic Value', 75(2) *The Monist* 119 (1992). In the words of Brown (above n. 44, at 231): 'Economic measures of value are species of the genus assigned value, which belongs to the family value'. There is indeed a fundamental distinction between held and assigned values that Brown describes in depth (above n. 44, at 233). Held values refer to principles and ideals that are important to people; they can be instrumental values, such as generosity or courage, and terminal values, such as happiness and freedom. There is a large body of literature, especially in psychology, on held values and how they may influence human behaviours and environmental concerns. For instance, held values have been grouped into clusters (anthropocentric, ecocentric, egoistic, socio-altruistic, etc.) and clusters give a certain orientation to human values for the environment. *See* for more references on held values: E. Seymour, A. Curtis, D. Pannell, C. Allan & A. Roberts, 'Understanding the Role of Assigned Values in Natural Resource Management', 17 *Australasian Journal of Environmental Management* 142 (2010). Yet, held values do not say anything about social preferences for specific natural resources or particular changes in environmental quality (K. Segerson, 'Valuing Environmental Goods and Services: An Economic Perspective', in P.A. Champ, K.J. Boyle & T.C. Brown (eds.), *A Primer on Nonmarket Valuation* (2017) 1, at 6). Conversely, assigned values express the relative importance of an object to a group or individual in a given context, by implicit or explicit comparison (Brown, above n. 44, at 232). Therefore, economic valuation techniques developed over the past four decades focused on assigned values because they enable the understanding of how people trade-off environmental values (within the rational choice theory). *See* M.A. Freeman, *The Measurement of Environmental and Resource Values: Theory and Methods* (1993). For more references and discussion about assigned values, *see* Segerson (in this footnote, at 9ss).

[47] N. Hanley, 'The Economic Value of Environmental Damage', in M. Bowman and A. Boyle (eds.), *Environmental Damage in International and Comparative Law* (2002), at 27. Environmental values can be also measured through (monetary) impacts on production, i.e. through the impact of environmental changes on productive factors and, in turn, on profits. Yet, environmental damage assessment techniques mainly focused on the loss of individual utility and, thus, the measure of production losses is not taken into account in this article.

[48] However, the utility theory of value is just one of the possible approaches to values, which draws on the basic idea that values are given by the interaction between individual preferences and productive abilities. Another possible approach would be to measure values through the labour needed to produce goods (this is the typical approach in classical economics).

[49] For a short history of the utility theory applied to the environment in the western belief system, *see* S. Parks & J. Gowdy, 'What Have Economists Learned about Valuing Nature? A Review Essay', 3(C) *Ecosystem Services* e-01 (2013).

individual utility can increase or decrease if the quality of the environment changes. If an accidental event pollutes a beach, visitors will not be able to swim and they will see their utility reduced as a consequence of the accident. This change in utility can be regarded as a measurement of the environmental value and, namely, of the 'use value' of the environment. However, even people not using natural resources might suffer a loss of utility due to the accident. This is because we value our future possibility of using that environment or we care about the fact that future generations will benefit from the same possibility. More precisely, economists refer to these as 'non-use' or 'passive-use' values of environmental goods and services.[50] Drawing on this wider approach, in 1985 Boyle and Bishop laid the foundation of the concept of total economic value (TEV) of the environment.[51] Figure 1 below provides an easy-to-read taxonomy with some examples:

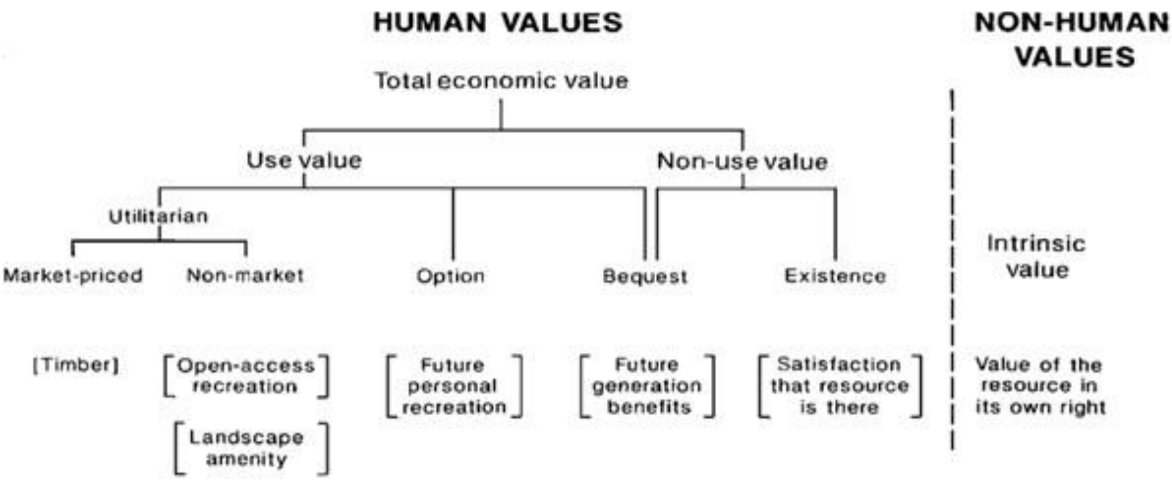

Figure 1 [The total economic value of nature][52]

As can be seen above, the TEV includes both 'use values' and 'non-use values' (or passive-use values) within the category of human values for the environment. Use values[53] are based on

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[50] Some economists keep criticising passive-use values by questioning their existence as well as the need for special assessment techniques (the so-called 'contingent valuation', *see* infra). Nevertheless, environmental policies to preserve natural resources void of use values (e.g. the Amazon rain forest) reveal the relevance of non-use values.

[51] K.J. Boyle & R.C. Bishop, 'The Total Value of Wildlife: A Case Study Involving Endangered Species', 278711 *1985 Annual Meeting, August 4-7, Ames, Iowa, American Agricultural Economics Association*, (1985).

[52] Source: I.J. Bateman, A.A. Lovett & J.S. Brainard, Applied Environmental Economics. A GIS Approach to Cost-Benefit Analysis (2003), at 2.

[53] The 'use value' differs from the 'exchange value'. The former relates to the benefit of using natural resources independently from the fact that they are traded in the market. The latter (exchange value) is basically the price or the commercial value. Say and Ricardo were the first scholars who, in the beginning of the 19th century, pointed out that natural resources may have a high use value even if they have no exchange value (price). Neoclassical economists in the 20th century further emphasised use values. This distinction explains the apparent paradox of goods with a high use value and a very low exchange value (e.g. water) and goods with a low use value and a very high exchange value (e.g. diamonds). For a historical overview of economic schools of thoughts on the value of natural resources, *see* E. Gómez-Baggethun, R. de Groot, P.L. Lomas & C. Montes, 'The

the actual, future or possible use (option value) of environmental goods, whereas non-use values[54] refer to the social preference for the mere existence (existence value)[55] or for the possible/actual use from future generations (bequest value). Intrinsic values, which are independent of human preferences, are by definition not encompassed by the TEV, although they may influence non-use values.[56]

Within this traditional framework, the next step is to understand how to assess the various values. Since in neoclassical economics values are linked to utility, valuation techniques aim to measure utility changes. Let us now assume that individuals enjoy the same level of utility when a reduction in the quantity of one good is compensated by an increase in the quantity of another good, that may be anything but in practice it is often money.[57] The obvious consequence of this common assumption is that a measure of the trade-off between the object of valuation and something else in exchange can be regarded as the 'true value' of the good whose value needs to be assessed. With environmental changes, the problem is that it is often impossible to directly infer their value from market prices. How to measure the value of a polluted beach after an oil spill if there is no market price to look to? Environmental goods that are not bought and sold in the marketplace, such as beaches, wildlife, rivers and fresh air, are known in economics as non-market goods[58] and the tools developed to measure their value are

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History of Ecosystem Services in Economic Theory and Practice: From Early Notions to Markets and Payment Schemes', 69(6) *Ecological Economics* 1209 (2010).

[54] The origins of this notion date back to the end of the 1960s. In 1967 John Krutilla published the paper titled 'Conservation reconsidered' in the *American Economic Review*. His aim was to bring about a change in the field of conservation economics by shifting the traditional focus to natural areas that were not efficiently provided by the market and they thus risked being underprovided in the future (e.g. national parks). From the perspective of Krutilla, these amenities needed to be protected in spite of missing use values but in view of their future recreational value. He never talked about a total economic value but his lesson is deemed as foundational in the field of modern environmental economics. The development of non-market valuation techniques to measure passive-use values exploded in the years that followed his paper. *See* J.V. Krutilla, 'Conservation Reconsidered', 57(4) *American Economic Review* 777 (1967). In 2003 Freeman defined non-use values more broadly as all values that are not measurable by revealed preference methods; in this way difficulties in defining what is 'use' are avoided.

[55] The existence value means that people gain utility from knowing that a natural resource exists even if the individuals expressing their values have no actual or planned use for themselves or anyone else. Therefore, they would be willing to pay for its preservation.

[56] In this article, we do not enter into the debate on intrinsic values and how to account for them. Suffice it to say, the notion has been mainly discussed in the philosophical literature rather than in economics. Indeed, it is much unclear from the perspective of the utility theory how people would trade off intrinsic values with other values. For this reason, the TEV traditionally does not include intrinsic values, but it is possible to elicit them through stated preference methods.

[57] This is a basic assumption in the utility theory of value and in line with the rational choice theory. *See* R.C. Bishop & K.J. Boyle, 'Reliability and Validity in Nonmarket Valuation', in P.A. Champ, K.J. Boyle & T.C. Brown (eds.), *A Primer on Nonmarket Valuation* (2017) 463, at 465.

[58] There are a lot of goods falling in the category of environmental goods: air quality, water quality, amenities such as a good view on nature, etc. Environmental economics includes in this category everything for which people may have preferences. They differ from ordinary goods because there is no market for them and, thus, it is not easy to build a demand curve and deduce their value from the interaction between demand and supply. They belong to the larger category of public goods (goods that are non-rival, i.e. they can be simultaneously consumed by everyone, and non-excludable, i.e. nobody can be excluded from consuming them by, for instance, paying a price). *See* C.D. Kolstad, *Environmental Economics* (2000), at 289ss.

called non-market valuation techniques. Their goal is to measure the 'true value' for a change in the quality of environmental goods and services.[59]

Before introducing them, why they were developed needs to be clarified. According to Segerson, the first techniques to value natural resources in the US appeared in the 1950s and they were used by federal agencies in benefit-cost analyses of water projects, such as dam constructions.[60] In the years that followed economists further refined and improved those techniques, since new laws, such as the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) of 1980[61] and other regulations, required either to estimate compensation for damage after environmental accidents or to assess costs and benefits of environmental policies.[62]

Having said that, we can now go back to the practical valuation of non-market goods when they are not traded. Absent prices, environmental economists developed similar concepts equally applicable to environmental goods in order to measure their demand curve: the maximum amount of income that an individual would be willing to give up in order to have more of another good and keep the same utility level as before (compensating welfare measure or willingness to pay, WTP)[63] and the (minimum) amount of additional income that an individual would need to gain in order to give up something that he already owns and keep the same utility level as before (equivalent welfare measure or willingness to accept, WTA).[64] Which one to use depends on the assignment of property rights. An example can be useful. Imagine that we want to assess the value of an environmental loss caused by an accident. The ex ante WTP is the maximum amount of money that individuals would be willing to give up for introducing measures that avoid the occurrence of accidents (and related losses) and for

[59] Bishop and Boyle define the 'true value' or WTP for a change in environmental quality, as 'the maximum income that a consumer would be willing to give up to have the same utility as before after the environmental change takes place'. *See* Bishop and Boyle, above n. 57, at 465.

[60] Segerson, above n. 46*,* at 4.

[61] The CERCLA and the Oil Pollution Act (OPA) of 1990 triggered the improvement of non-market valuation techniques because they allowed victims of accidents to sue for damage compensation.

[62] The cost-benefit analysis is a popular technique aimed at identifying, quantifying and weighing the costs and benefits of projects and policies, including the environmental impacts (costs and benefits).

[63] In principle, the good used as term of reference could be anything. In practice, economists have generally used money to measure values.

[64] Much attention in the economic scholarship revolved around the difference in size between the two measurements, given by the fact that the WTP is bound by income (it is influenced by the income of the valuator), that people value losses more than gains because they are more willing to pay to maintain their status quo rather than paying to improve it (prospect theory). *See* D. Kahneman & A. Tversky, 'Prospect Theory: An Analysis of Decision under Risk', 47(2) *Econometrica* 263 (1979). Moreover, the absence (or scarcity) of good substitutes for environmental quality might bring to a higher WTA compared to the WTP, because people would ask more money to accept a higher risk of degraded environment rather than what they would be willing to pay for a reduced risk of it. For a deeper understanding of all these issues, *see* W.M. Hanemann, 'The Economic Theory of WTP and WTA'', in J. Bateman & K.G. Willis (eds.), *Valuing Environmental Preferences: Theory and Practice of the Contingent Valuation Method in the US, EU, and Developing Countries* (2001), at 42.

keeping their utility as before the accidents, whereas the WTA is the minimum money that individuals would be willing to accept in order to tolerate a lower value of the environment. Whether to adopt the WTA or the WTP depends on the entitlement prior to the accident: if people had the right to enjoy a pre-loss level of utility from the environment, then it would be appropriate to measure the WTA.[65] But, how to measure the WTP (or the WTA) in practice? The next section will explain in more detail which techniques of non-market valuation have been developed to assess use and non-use values of the environment.

5. The methods of nature valuation in environmental economics

As stated earlier, environmental goods and services are usually not traded in the marketplace. Indeed, it rarely happens that goods, like timber or fruits, can be bought and sold. Only in these relatively few cases, it is possible to elicit the value of the environment from prices. This type of valuation technique is thus called market-based. If instead there is no market price for natural resources, then it is necessary to resort to non-market valuation methods. The methods of non-market valuation in environmental economics are grouped into two main categories: revealed and stated preference. Revealed preference methods indirectly imply values from observed behaviours in surrogate markets (e.g. house market) or existing markets (e.g. how many people buy the ticket to visit a park), whereas stated preference methods directly extract the maximum WTP or the minimum WTA from answers to survey questions (hypothetical market). The main difference between the two classes is not only the technique, but also the components of TEV which they can capture. Revealed preference methods only capture use values, while stated preference techniques are ideally able to capture both use and non-use values. However, each existing method captures use or non-use values limited to a specific category of goods (e.g. hedonic pricing only looks at goods with a price, such as houses). Figure 2 on the next page provides a synthesis of the relationships between TEV, methods and proxies.

[65] E.S. Goodstein and S. Polasky, *Economics and the Environment* (2004), at 78. The authors explain that if people think that clean air or clean water belong to them, then the value for a reduction of environmental quality would be better expressed by the willingness to be compensated for their degradation. For this reason, survey studies should correctly measure the WTP for private goods and the WTA for common goods.

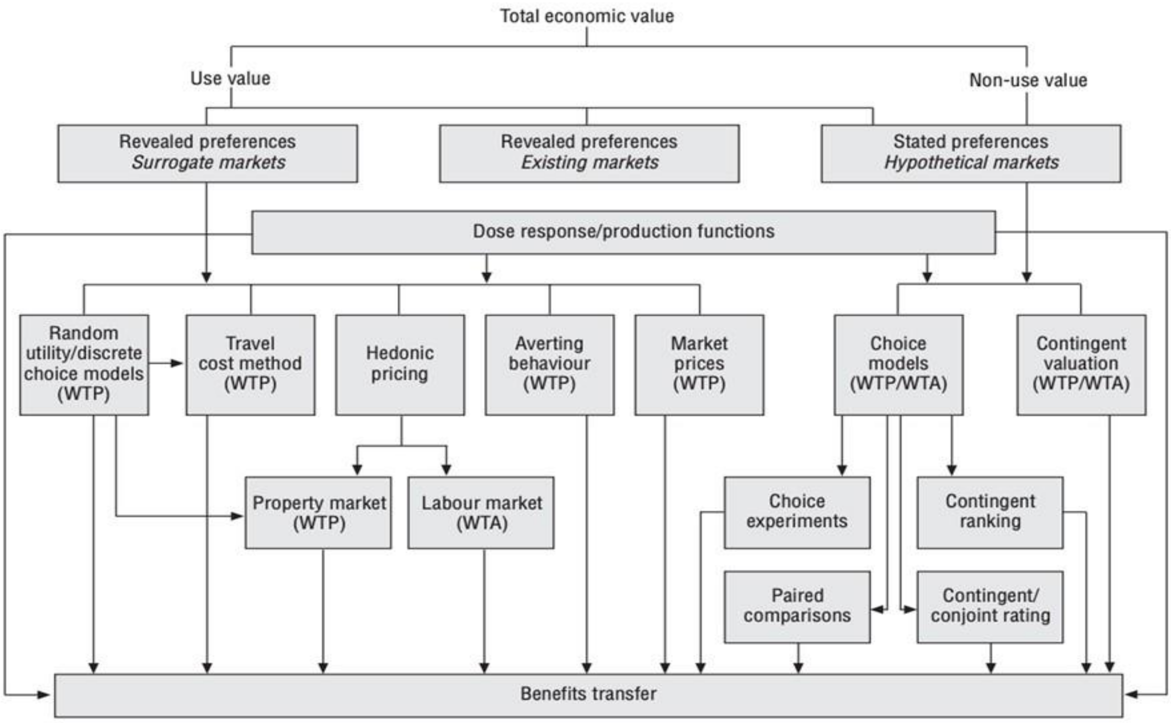

Figure 2 [Total economic value][66]

Some observations based on the figure above are needed. Non-use values, which are highly relevant for natural resources with few or no substitutes (e.g. a unique natural place), can only be estimated through stated preference methods (questionnaires). Revealed preference methods cannot elicit non-use values for the simple reason that non-use values are not linked to behavioural changes[67] in the marketplace (e.g. a change in the demand or the supply). Whether a valuation method is likely to elicit both use and non-use values of the environment is pretty relevant from a perspective of law and economics. Let us assume that an environmental good has been damaged and it held a huge non-use value compared to the use value (e.g. a natural area used not used either for recreation or for other goals). In this case, neither prices nor revealed preference would capture its total value. As a consequence, liability laws are expected to send to potential polluters wrong incentives of precaution (and activity), hence causing underdeterrence and pollution beyond optimal levels. A stated preference method would instead allow to obtain estimates that should be closer to the 'true value'[68] of the lost environment, provided that questionnaires have been properly designed to ensure reliable and

[66] Source: D. Pearce, G. Atkinson & S. Mourato (2006), above n. 10, at 88. Under this framework, production functions play a central role because there is a link between policy change, a change of the environment and some responses. For instance, a change of air quality (dose) would bring about a response in the number of sick people (output). Therefore, production functions should be taken into account to determine the TEV.

[67] In the words of Pearce and Mourato, a 'behavioural trail' (*ibid.,* at 86).

[68] *See* above n. 59.

accurate answers (see Section 5.3.2). Therefore, the latter methods should be preferred if one wishes to internalise the full cost of environmental accidents.

In addition, the method of benefits transfer involves the application of available estimates (from other studies) to natural resources with similar characteristics. Its validity has been highly debated and it can be considered valid under certain circumstances (similar environments, reliable and accurate estimates). Given than benefits transfer is built upon other methods is commonly known as a 'secondary valuation method' that relies on primary estimates from stated and revealed preference methods.

In addition to use and non-use values, further issues need to be taken into account in view of minimising the total social costs of accidents, such as the reliability, the validity and the same costs of valuation. A central goal in the valuation of the environment is indeed to produce accurate value estimates. Reliability and validity are the common criteria of accuracy in environmental economics. Reliability has to do with variance and erratic results, whereas validity refers to unbiased results. These concepts will be clarified in the following subsections which briefly illustrate the advantages and shortcomings of each category of valuation techniques. Within each category, the focus will be on the main methods that have been employed by judges in liability cases, rather than tackling all the existing non-market valuation techniques. For this reason, methods like choice models and averting behaviour will be only briefly mentioned.

5.1 Market-based approaches

When environmental goods and services can be traded in markets, such as fruits and timber, it is possible to infer their values directly from market prices. To be more precise, market-based approaches may look at either the cost side or the benefit side.[69]

Cost-based valuation is based on the assumption that expenditures on producing and maintaining environmental goods or services provide net benefits. It requires the elaboration of hypothetical scenarios that respond to the question: what would be the cost to bear if the environmental good or service had to be artificially recreated? Figure 3 on the next page illustrates the cost side of market-based approaches.

[69] This is a traditional classification from IIED (International Institute for Environment and Development), 'Economic Evaluation of Tropical Forest Land Use Options: A Review of Methodology and Applications' (1994).

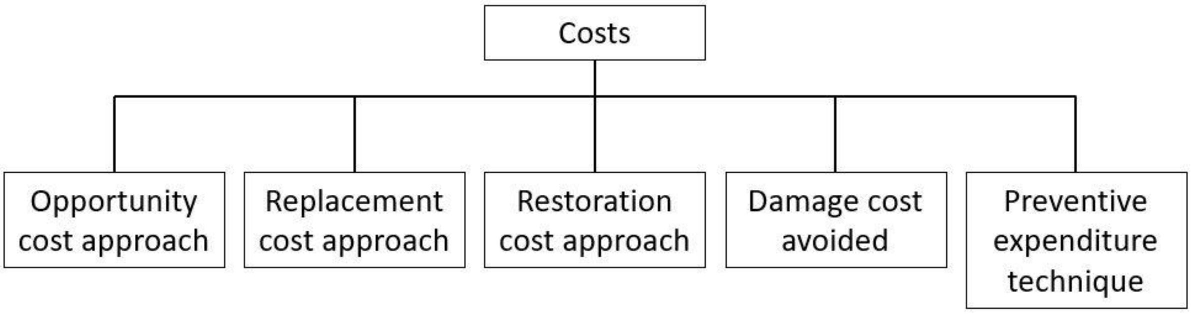

Figure 3 [Taxonomy of market-based valuation techniques][70]

The opportunity cost approach derives from the idea that the opportunity cost of unpriced uses (e.g. forest conservation) can be inferred from the foregone income of other uses (e.g. forestry).[71]

The replacement cost approach looks at the expenditures incurred to replace the impaired natural resources with substitutes. The underlying idea is that replacement costs provide a measure of the minimum WTP to keep receiving a certain benefit (assuming that individuals have correct information about the damage).

The restoration cost approach, like the preventive expenditure method, estimates the cost of activities to maintain a certain level of enjoyment or output, including the relocation of individual activities, households and firms or adjustments to maintain an activity in the existing location.

The damage cost avoided infers the value of the environment from the costs incurred to avoid the environmental damage. Yet, not all agree that the damage cost avoided is a cost-based approach because it is based on the assumption that the cost of damage is a measure of value.[72]

The preventive expenditure technique or mitigation cost approach looks at the costs that households are willing to pay to prevent future environmental damage and keep stable their existing level of utility. Presumably, individuals are willing to spend up to the point where the costs equal the benefits derived from a protected environment. Their WTP can be then inferred through stated preference (contingent valuation or CV) or revealed preference (from similar events in the past).

[70] The table can be found in the notes prepared by A.N.A. Ghani for the lecture on 'Market-based Techniques', at 4. *See* www.blogs.ubc.ca/apfnet04/module-5/topic-1-market-based-techniques/.

[71] For instance, the time spent harvesting may be valued in terms of foregone rural wages (opportunity cost of labour). *See* E.B. Barbier, M. Acreman & D. Knowler, 'Economic Valuation of Wetlands. A Guide for Policy Makers and Planners', Ramsar Convention Bureau, at 42 (1997). Note that the information about opportunity costs can be then obtained also through stated or revealed preference (hypothetical or surrogated markets).

[72] Barbier et al., above n. 71, Appendix 3, at 11.

Market-based valuation techniques from the benefit side look instead at the market value (price) or the change in income of productive factors. The underlying rationale for using prices is that if natural resources physically contribute to the production of other commodities or services traded in markets (e.g. fishing, hunting and farming), changes in ecological functions (improvement or deterioration of environmental quality, e.g. water quality) may affect the quantity or price of certain goods.[73] On the other hand, changes in income can be used to measure the value of the environment. Indeed, environmental pollution may be the cause of sicknesses, premature death, increased medical expenses that, in turn, lead to a diminution in the workers' income.

5.1.1 Advantages

Market prices are usually considered to provide accurate information on the value of natural resources since they embed market preferences and marginal costs of production, which means data from actual markets. This may have three well-known advantages. First, data on prices, quantities and costs are easy to obtain, less resource-intensive and not expensive. Secondly, market prices reflect the actual WTP for costs and benefits that are traded, so they are considered to be sufficiently accurate to reflect the 'true value' of nature. Thirdly, these data are generally regarded as sufficiently objective and thus more reliable than other tools to elicit social preferences.

5.1.2 Limitations

The main limitation of these approaches is that they are applicable to the extent that markets exist and data on prices or costs are available. More often, choices on environmental goods and services are not observable in market transactions because they are public goods and usually not traded in markets.[74] For instance, even if we value bats and we would be willing to pay for their conservation,[75] there is no market where we can express our preference for their preservation.

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[73] A.M. Freeman, 'Valuing Environmental Resources under Alternative Management Regimes', 3(3) *Ecological Economics* 247 (1991). Also: A.M. Freeman, J.A. Herriges & C.L. Kling, *The Measurement of Environmental and Resource Values. Theory and Methods* (2003)*,* at 259.

[74] In economics, public goods are those commodities or services which are available for the whole society, non-excludable (there is no technology available to exclude others from using the same good) and non-rivalrous (individual consumption does not reduce the quantity available for others). The fact that we breath air does not exclude others from breathing and does not consume the quantity available for the others.

[75] It might be interesting to know that the fate of bats has been at the forefront of a recent case before the Hawaii Supreme Court due to a contentious wind farm. According to the plaintiffs, a local community for which bats hold cultural and spiritual

The second limitation is that, even if market prices are available, they might be distorted by policy interventions (e.g. subsidies or taxes),[76] monopolies,[77] seasonal variations, etc. This limitation can be overcome by adjusting prices (so-called 'efficiency shadow prices method') so that they reflect the true WTP. Yet, shadow pricing might face further criticism due to the artificial nature of data, the fact that it is based on assumptions and it might suffer from inaccuracies.[78]

Last but not least, prices only refer to the preferences of those who use non-market goods and with whom there is a clear demand link (see above the distinction between value of use and value of exchange). However, there are cases when the demand is unidentifiable and this does not mean that people do not value non-market goods.[79] Simply, market-price approaches cannot capture non-use values by those who do not use environmental goods (so, there is no demand link) but would still be willing to pay for their conservation or improvement.

Further limitations specifically relate to some approaches. For instance, Barbier warned that the replacement cost method should be used with caution because it is unsure whether the benefits of the replacing resource are equal to the benefits of the original damaged resource if data on the original ecological functions are not available.[80] Moreover, Daily pointed out that direct relationships between resources and economic outputs are often difficult to estimate.[81] Additional issues of inaccuracy are given by the fact that restoration costs might exceed the benefits of the original resources if data on the baseline are missing and/or restoring previous conditions might be difficult. Likewise, it is unlikely that relocated environmental commodities can provide the same benefits of the lost ones in the original location.[82]

values, the windfarm project did not follow the standards set by the law to protect endangered species, hence causing the death of 51 bats per year. How to weigh the social benefit of a windfarm with the social cost represented by the ecological and cultural loss of 51 bats per year if they have no price? For more details, see this short commentary with useful references: www.jindalsocietyofinternationallaw.com/post/bat-fatalities-at-kahuku-windfarm-making-a-case-under-international-environmental-law.

[76] It is quite well known that subsidies distort market prices and they thus interfere with the conduct of economic agents. Technically, subsidies can reduce the marginal costs of recipients or raise their marginal revenues. In this way, subsidies provide the ability to produce at lower costs, so that recipients enjoy a competitive advantage and they can increase the production. As a consequence, prices might inefficiently increase. An exception is given by subsidies for Research and Development (R&D). This category of subsidies addresses a typical market failure, since the provision of knowledge created by programs of R&D is publicly available. For this reason, the private revenues would not equal the costs and its provision would be lower than efficient. *See* R. Diamond, 'Privatization and the Definition of Subsidy', 11 *Journal of International Economic Law* 649 (2008).

[77] Monopolies without government interventions lead to higher prices and a consumer welfare lower than efficient levels (more welfare for the monopolistic producer).

[78] A. Smith, 'Shadow Price Calculations in Distorted Economies', 89(3) *The Scandinavian Journal of Economics* 287, at 302 (1987).

[79] N.E. Flores, 'Conceptual Framework for Nonmarket Valuation', in P.A. Champ, K.J. Boyle & T.C. Brown (eds.), *A Primer on Nonmarket Valuation Second Edition* (2017), at 44.

[80] Barbier et al., above n. 71, Appendix 3, at 10.

[81] G.C. Daily, 'Ecosystem Services: Benefits Supplied to Human Societies by Natural Ecosystems', 2 *Issues in Ecology* 1 (1997).

[82] Barbier et al., above n. 71, Appendix 3, at 10.

5.2 Revealed preference methods

When prices of environmental goods and services are not available, but there are markets closely related to them, revealed preference methods can be applied. These techniques are based on the observation of preferences shown, i.e. 'revealed', in actual market transactions which have a correlation with the natural resource to value. Two main methods are used to elicit revealed preference: travel cost models (TCMs) and hedonic pricing (HP).

TCMs are used to value recreational uses of natural resources, such as fishing, rock climbing, boating, swimming and hunting.[83] The underlying insight is that the cost of the trip to reach a site corresponds to the individual's price for recreation (lower bound). Therefore, individuals reveal their WTP for recreation through the number of trips they do and the site they choose to visit. Changes in the demand function for recreation can indeed provide a measure of changes in preferences for the quality or quantity of environmental goods and services. The use of TCM has been largely motivated by the need to conduct benefit-cost analyses of environmental regulations or for damage compensation after accidents.[84]

HP is used to estimate the implicit prices of characteristics over heterogeneous or differentiated products (distinct varieties of one product).[85] Imagine that a product is sold in one market but

[83] The earliest travel cost models date back to the 1950s and they followed the method proposed by Hotelling. They measure visitation rates for geographic zones defined around single recreation sites. *See* H. Hotelling, 'An Economic Study of the Monetary Valuation of Recreation in the National Parks, Washington', US Department of the Interior (1949).

[84] 'Economists have been concerned with measuring the economic value of recreational uses of the environment for more than 50 years' (G.R. Parsons, 'Travel Cost Models', in P.A. Champ, K.J. Boyle & T.C. Brown, *A Primer on Nonmarket Valuation Second Edition* (2017), at 187ss). Most research in the 1960s aimed at valuing per-trip values in order to support conservation versus development of large water resource projects (at least in the United States). In the late 1970s and in the 1980s, the interest moved to valuing quality changes at recreation sites induced by policies willing to improve the quality of the environment. In the 1980s much research was conducted on beach uses and recreational fishing in Alaska. *See* N.E. Bockstael, W.M. Hanemann & I.E. Strand, 'Measuring the Benefits of Water Quality Improvements Using Recreation Demand Models', Report presented to the US Environmental Protection Agency. College Park: University of Maryland (1984); N.E. Bockstael, M.W. Hanemann & C.L. Kling, 'Estimating the Value of Water Quality Improvements in a Recreational Demand Framework', 23 *Water Resources Research* 951 (1987); R.T. Carson, W.M. Hanemann & T.C. Wegge, 'Southcentral Alaska Sport Fishing Study', Report prepared by Jones and Stokes Associates for the Alaska Department of Fish and Game, Anchorage, AK (1987); R.T. Carson, W.M. Hanemann & T.C. Wegge, 'A Nested Logit Model of Recreational Fishing Demand in Alaska', 24 *Marine Resource Economics* 101 (2009). Economists started to look at many more recreational activities (fishing, swimming, boating, climbing, hiking, hunting, skiing, etc.). During the past two decades, models have been further improved and refined. The latest models (Kuhn-Tucker) try to integrate seasonal and site choices into a unified utility framework.

[85] L.O. Taylor, 'Hedonics', in P.A. Champ, K.J. Boyle & T.C. Brown (eds.), *A Primer on Nonmarket Valuation Second Edition* (2017), at 235. Although popularised by Griliches in the 1960s, the coining of the term 'hedonic' dates back to a 1939 article by Andrew Court. Court was an economist working for the Automobile Manufacturers' Association in Detroit from 1930 to 1940. Examining automobile prices indices, he noticed that passenger cars serve so many different uses that one single most important characteristic cannot be identified. Therefore, prices cannot be compared by applying a simple regression method. He proposed instead to employ single composite measures. In his work, hedonic specifically refers to an index of 'usefulness' that combines the relative importance of various characteristics (braking capacity, horsepower, etc.). Hedonic indexes can be then compared. For a description of Court's work, *see* A.C. Goodman, 'Andrew Court and the Invention of Hedonic Price Analysis', 44 *Journal of Urban Economics* 291 (1997). In his words: 'Hedonic price comparisons are those which recognize the potential contribution of any commodity, a motor car in this instance, to the welfare and happiness of its purchasers and the community' (*ibid*., at 292, footnote 2).

characteristics vary in such a way that there are distinct product varieties. It is possible to indirectly observe the monetary trade-off which individuals are willing to make by observing the difference in price between two product varieties which vary only by one characteristic (e.g. two identical houses, but one has an additional room).[86] For this reason, HP is an indirect valuation method that infers values from observable market transactions. In the environmental domain, it is commonly applied to the housing market. Let us take an example. If there are two identical houses in front of two different lakes (one with improved water clarity), the price differential determined by the increasing demand for the house in front of the lake with better water is the implicit price consumers are willing to pay for that environmental amenity (water clarity). Implicit or hedonic prices allow therefore to elicit the WTP for that specific environmental component.

5.2.1 Advantages

The first advantage of revealed preference methods is that there is broad agreement among researchers on the steps that need to be followed to achieve minimal accuracy in estimating true values. The TCM is considered to be a high-ranking tool among revealed preference techniques and there is widespread confidence on its validity,[87] whereas HP is one of the most popular methods thanks to the minimal data requirements and its easy empirical implementations.[88] Scholars emphasise the existence of a clear procedure that starts from the search of a surrogate market close to the environmental goods and services to be valued. The procedure follows with the choice of the appropriate method (TCM or HP). Then, the needed data are collected according to the relative procedures[89] in order to build the demand function.[90] Subsequently, the value of a marginal change in the quality or quantity of environmental good is deducted from the demand function. Lastly, values are aggregated and discounted. For the HP, information on sales prices is always readily available, with considerable savings of time and costs. Moreover, data acquisition costs have been decreased, hence making both stages of HP cheaper.[91]

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[86] The utility theoretic framework needed to build the demand function for characteristics of heterogeneous products has been developed by Rosen in a seminal paper. *See*: R. Rosen, 'Hedonic Prices and Implicit Markets: Product Differentiation in Pure Competition', 82 *Journal of Political Economy* 34 (1974).

[87] Bishop and Boyle (2017), above n. 57, at 489.

[88] Taylor, above n. 85, at 285.

[89] In TCM, recreation surveys are designed, sent around and analysed according to a precise step-wise guide (Parsons, above n. 84, at 203). In HP, there are two subsequent steps: collection of marginal price information and then estimation of the demand function by combining information on prices and data on household characteristics (Taylor, above n. 85, at 237).

[90] A typical set of questions in TCM surveys is: 1) trip count and location; 2) last trip: 3) stated-preference question; 4) respondent and household characteristics.

[91] Taylor, above n. 85, at 285.

5.2.2 Limitations

There are various limitations to revealed preference methods. Studies on TCM have been much concerned with accuracy issues, starting from the 1960s.[92] Yet, such research has never been explicitly revolving around the topics of reliability and validity.[93] Apparently, Bishop and Boyle made a first attempt in this regard and they argued that their conclusions can also be applicable to other revealed preference methods.[94] Regarding reliability, it seems that: 'using recreation-participation data with long periods of recall could tend to increase the variance of reported participation and hence reduce the reliability of the travel cost method, all else being equal'.[95] In other words, the time of recall (i.e. the time to reconstruct the behaviour on which respondents to surveys are supposed to report) might make the method less reliable with long recall periods. Therefore, for reliable data it is essential to ensure short recall periods. As to the validity side, there are still a number of partially unresolved issues that have been not directly addressed. Parsons identifies a list of 'soft spots' that need to be improved in TC modelling, such as the current way of measuring time, overnight trip modelling, multipurpose trips, integration of site choices with trip frequency, the inclusion of more welfare-revealing choices, the error introduced by the recall bias and, finally, more integration with stated preference studies.[96] Another important aspect is that most of the research on TCM ignores dynamics in decision-making (intertemporal substitutions) that would allow people to substitute sites over time or to base current decisions on expectations about future trips. Most models consider instead individual trip choices day by day over a season independently of decisions on future trips. Consideration of interdependencies between different trip choices would indeed require more complex ways of gathering data, more surveys and, in general, higher costs.[97] Furthermore, trip costs are considered to be given but they can be also the result of subjective choices.[98] Another issue is the 'recall bias', occurring when people report visiting sites more frequently than they actually do. The validity of the TCM might be considerably reduced by all these issues. In order to offset possible biases and ensure validity, it is important to carefully follow all the well-established steps of the method and to clarify all the assumptions in advance.

[92] Bishop and Boyle (2017), above n. 57, at 487.
[93] As already said above at footnote n. 13, reliability and validity are criteria to assess the accuracy. Reliability has to do with variance and erratic results, whereas validity refers to unbiased results.
[94] Bishop and Boyle (2017), above n. 57, at 487.
[95] Ibid., at 488.
[96] Parsons, above n. 84, at 225.
[97] For instance, people should receive reminders to respond to several seasonal surveys.
[98] For instance, current models use the behaviour of those with higher travel costs in order to predict the behaviour of those with lower costs in case the price of visits increases. Yet, people might choose to live closer to a recreational site and this approach would underestimate their preferences. *See* Bishop and Boyle (2017), above n. 57, at 489.

It has been also warned in the literature that travel cost studies may give higher values than stated preference studies, hence raising the need for more research on convergent results.[99] In addition to the limitations related to accuracy, revealed preference require the existence of surrogated markets and, if data are not already available, the process of gathering good-quality data might take time and costs. Lastly, it needs to be considered that revealed preference cannot capture non-use values and, thus, the total value of natural resources with high non-use would not be accurate (even on average).

5.3 Stated preference methods

Stated preference approaches are based on surveys that try to elicit social preferences about policies that may change the provision of natural resources. Three types of techniques fall in this category. The most popular methodology is CV, where people are asked how much money (maximum) they would be willing to spend in order to increase the provision of environmental goods or services or, alternatively, how much money (minimum) they would need to receive in order to be willing to accept their loss. The second popular method is choice modelling (CM), which tries to model the decision process of individuals in the face of two or more alternatives about the goods or services to value.[100] Lastly, group valuation combines stated preference techniques with deliberative processes from political sciences in order to capture components of values others than those elicited through surveys.[101]

5.3.1 Advantages

Stated preference methods of valuation ideally allow to directly elicit preferences about the values of natural resources and to obtain the best theoretical measures of WTP or WTA. Moreover, these are the only techniques to estimate non-use values (option and existence values) and estimate the TEV. Furthermore, a CM study allows to derive marginal values for changes of specific attributes of environmental resources induced by different policies (options). Each option in the survey consists indeed of a different balance of impacts on the environment, such that choosing one option rather than the other reveals preferences about a

[99] See Bishop and Boyle (2017), above n. 57, at 491.

[100] The main difference between contingent valuation (CV) and choice modelling (CM) is that in a CV respondents have only one option and they are asked whether they would agree on paying for it or they would rather stick to the *status quo*, whereas in a CM study respondents are given several choices.

[101] Spash refers to value pluralism, incommensurability, non-human values and social justice. C.L. Spash, 'How Much is That Ecosystem in the Window? The One with the Bio-diverse Trail', 17 *Environmental Values* 259 (2008).

specific change of attributes. Differently from the other techniques, group valuation has the potential of overcoming limitations of traditional monetary valuation methods.[102] Lastly, Adamowicz pointed out how stated preference approaches turn out to be more useful than other methods because they provide information regarding perceptions, attitudes and previous knowledge.[103] All these additional pieces of information may help us to understand better preferences for the assessment. For instance, stated preference may show the relative importance given by respondents to different environmental services[104] as well as conflicts among stakeholders about alternative policy options.[105]

5.3.2 Limitations

Stated preference valuation methods raise several concerns in terms of accuracy (reliability and validity) which challenge the truth of the estimated WTP/WTA.

First of all, answers to survey questions depend on the way questions are designed and four main causes of errors might lead to biased answers: hypothetical bias (poorly thought out answers to questions that present events as mere possibilities), free riding (the belief that others will take on the responsibility of paying for public goods), strategic bias (the assumption that the stated answer will lead to adopt a specific environmental policy), embedding bias (error given by, for instance, the order of questions).[106]

Secondly, scholars stress the discrepancy between WTP and WTA.[107] It has been proved that the WTA is higher than the WTP for identical resources.[108] Various causes may explain this divergence: questionnaire designs, strategic behaviours and psychological effects, such as 'loss aversion' and the 'endowment effect'.[109] Another issue that may affect the validity of the estimates is the 'embedding bias',[110] or the fact that people tend to express the same WTP for

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[102] R. de Groot, M. Stuip, M. Finlayson & N. Davidson, 'Valuing Wetlands: Guidance for Valuing the Benefits Derived from Wetland Ecosystem Services', International Water Management Institute (2006).

[103] W.L. Adamowicz, 'What's It Worth? An Examination of Historical Trends and Future Directions in Environmental Valuation', 48 *Australian Journal of Agricultural and Resource Economics* 419 (2004).

[104] B. Martín-López, C. Montes & J. Benayas, 'The Non-economic Motives Behind the Willingness to Pay for Biodiversity Conservation', 139 *Biological Conservation* 67 (2007).

[105] P. Nunes, S. Silvestri, M. Pellizzato & V. Boatto, 'Regulation of the Fishing Activities in the Lagoon of Venice, Italy: Results from a Socio-economic Study', 80(1) *Estuarine, Coastal and Shelf Science* 173 (2008).

[106] Barbier et al., above n. 71.

[107] M. Hanemann, 'Willingness to Pay and Willingness to Accept: How Much Can They Differ?', 81(3) *American Economic Review* 635 (1991).

[108] V. Arild & D. Bromley, 'Choices without Prices without Apologies', 26(2) *Journal of Environmental Economics and Management* 129 (1994). *See* above section 5.

[109] K. Willis & G. Garrod, 'Valuing Landscape: A Contingent Valuation Approach', 37 *Journal of Environmental Management* 1 (1993).

[110] 'The embedding effect is the name given to the tendency of willingness-to-pay responses to be highly similar across different surveys, even where theory suggests (and sometimes requires) that the responses be very different'. *See* P.A.

an environmental change in a small area and in a bigger area because they are truly insensitive to the scope of the survey.[111] In any case, stating preferences about the environment is as challenging as valuing public goods for which preferences are not well defined and responses tend to lack sufficient accuracy.[112] Admittedly, upfront information in questionnaires[113] and valuation workshops held in advance[114] may help respondents to reflect on their preferences and overcome their cognitive constraints during surveys. Likewise, deliberative monetary valuation methods seem to further reduce biases and non-response rates, while raising the level of engagement of respondents.[115] Moreover, it is now possible to develop well-designed surveys to reduce the risk of error, although they might be highly expensive.[116]

The last fundamental limitation concerns the controversy still existing around the incommensurability of non-use values.[117] More specifically, the issue is whether non-use values (e.g. bequest values) can be put under the framework of the TEV together with recreational values and other economic values. The issue is still largely debated in the literature.

6. Comparing environmental valuation methods from the efficiency perspective

After reviewing the law and economics of damages and presenting the existing methods of environmental damage assessment from a theoretical perspective, it is now possible to compare and draw conclusions on their relative advantages and disadvantages.

First of all, there are four main issues of inaccuracy that may be common to all methods.

The first one relates to the relevant population whose values need to be estimated: should that be a limited group of people locally affected by the accident or the global population? If the aim of the valuation process is to compensate individuals for their post-accident losses, then it

Diamond and J.A. Hausman, 'Contingent Valuation: Is Some Number Better than No Number?', 8(4) *Journal of Economic Perspectives* 45, at 46 (1994).

[111] D. Kahneman & J. Knetsch, 'Valuing Public Goods: The Purchase of Moral Satisfaction', 22 *Journal of Environmental Economics and Management* 57 (1992); H. Svedsäter, 'Contingent Valuation of Global Environmental Resources: Test of Perfect and Regular Embedding', 21 *Journal of Economic Psychology* 605 (2000).

[112] H. Svedsäter, 'Economic Valuation of the Environment: How Citizens Make Sense of Contingent Valuation Questions', 79(1) *Land Economics* 122 (2003).

[113] C. Tisdell & C. Wilson, 'Economics of Wildlife Tourism', in K. Higginbottom (ed.), *Wildlife Tourism, Impacts, Management and Planning* (2004) 145.

[114] M. Christie, N. Hanley, J. Warren & K. Murphy, 'Valuing the Diversity of Biodiversity', 58(2) *Ecological Economics* 304 (2006).

[115] de Groot et al., above n. 102.

[116] Goodstein and Polasky, above n. 65, at 85.

[117] R. Carson, N.E. Flores & N. Meade, 'Contingent Valuation: Controversies and Evidence', 19 *Environmental and Resource Economics* 173 (2001).

makes sense to limit the assessment to the people affected by the accident and those legally entitled to compensation.[118]

The second issue concerns how individual values are aggregated. Normally, aggregated measures of benefits are not weighted based on the income, even if preferences expressed by wealthier people are higher compared to low-income people and this should be considered when interpreting the results of valuation processes.[119]

The third issue refers to the discount factor. The rationale for discounting is that people assign higher utility to immediate rather than future benefits (or they assign lower marginal utility to future benefits if an income increase is expected). Environmental policies pose an additional issue since future benefits are associated with future generations whose preferences should not to be weighted differently compared with present generations. The appropriate discount rate should thus depend on how utilities of different generations are weighted in a specific society and how consumption rates are expected to change over time.[120]

The fourth issue is the uncertainty of environmental changes over time and the fact that factors, like climate change, might change future outcomes. Uncertainties can be incorporated to increase accuracy by means of models that identify all possible scenarios and then assign probabilities based on risk attitudes. Yet, these models are highly resource-intensive and time-consuming.

Set aside the above-mentioned four causes of inaccuracy that may be common to all methods, the following three dimensions shall be inferred from the analysis above in order to compare the various valuation methods in terms of their efficiency:

1. accuracy (validity and reliability of estimates);
2. captured value of nature;
3. assessment costs.

Starting from accuracy, it is worth recalling the metaphor used by Bishop and Boyle to define it. According to these scholars, having accurate estimates is like 'shooting arrows at a target'.[121] Reliability can be understood in terms of arrows that are tightly grouped: this is the error

[118] Segerson (2017), above n. 46, at 15.
[119] Ibid.
[120] Increased consumption should bring to lower marginal utility in the future (*ibid.*, at 18).
[121] Bishop & Boyle (2017), above n. 57, at 464.

dispersion or the variance. On the other hand, validity has to do with the point on the target where arrows are centred: the distance from the centre is the bias or error. The figure below may help visualise the metaphor:

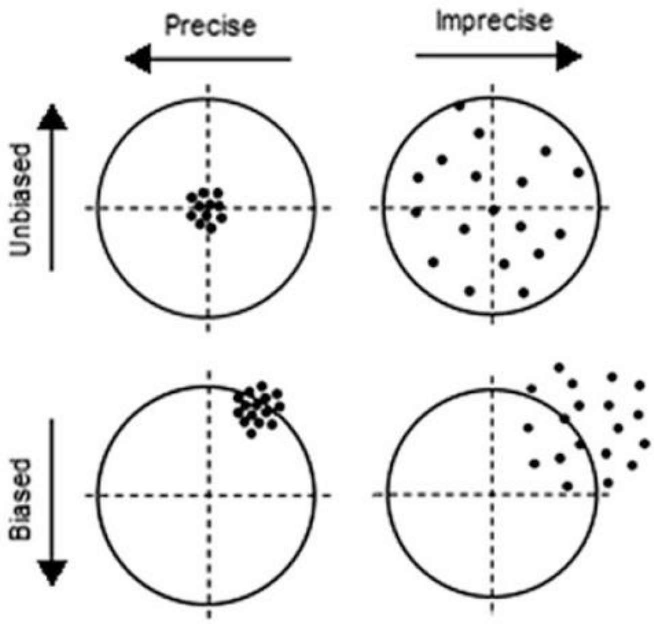

Figure 4 [Reliability and validity illustrated][122]

Accuracy embraces both aspects and estimates may be regarded as accurate if they are both reliable and valid. In the figure above, this situation is represented by the upper left circle.[123] This is where it is possible to find market-based approaches that are considered the most accurate. Revealed and stated preference methods can reach such a high level of accuracy only provided that very scrupulous procedures of assessment are conducted.

In terms of values of nature captured, market-based approaches and revealed preference only reflect use and exchange values, while stated preference also embed the values of those who do not use the natural resources in object but still gain utility from their existence.

When it comes to the assessment costs, market-based approaches are surely the least resource-intensive and cheapest tools, whereas stated preference techniques are more expensive due to the need of experts, time and money to run surveys and to process the answers. These costs can be considerably cut down only when studies on similar natural resources exist and their outcomes can be transferred to the damaged environment that has to be valued (benefits

[122] Ibid., at 466.

[123] All other circles refer to inaccurate results: unreliable but valid (upper right), unreliable and not valid (lower right) and reliable but not valid (lower left). For the indicators of reliability, see ibid., at 463.

transfer). However, this comes at the expense of accuracy and it is only possible between very similar ecosystems.

It is clear from the above that there is no ‘one size fits all’ solution for any kind of environmental damage assessment and that the most efficient method shall be determined based, on the one hand, on the specificities of the injured environment and, on the other hand, on the data and resources available.[124]

In the event of an environmental damage that is characterised by a major loss in use values, market-based or revealed preference approaches are more better placed to achieve optimal deterrence, provided that the necessary data (market prices) are readily available. This is because these approaches only encompass the use value of natural resources. As a consequence, potential polluters can easily retrieve this type of information and, thus, foresee their expected liability.

If, instead, the proportion of lost non-use values in relation to the total magnitude of environmental damage is considerable, market-based and revealed preference approaches should be eschewed since they fail to encompass non-use values and this would result in an underdeterrent effect for the polluter (see above §3). It is more socially desirable that stated preference approaches be employed in order to achieve better deterrence, particularly in the case of unique, irreplaceable and irrecoverable natural resources, where the non-use component of TEV is significant.

However, as already mentioned, the efficiency of methods also depends on the accuracy of the methods employed and the costs associated with the assessment process. Therefore, the employment of stated preference approaches is only efficient to the extent that:

a) they are rigorously applied, and
b) the non-use component of value is sufficiently high to outweigh the assessment costs.

Having said that, the last issue to tackle is whether the debate on the environmental damage assessment can be considered exhausted or instead something new might still contribute to change the way the environmental damage is valued in courts. The next section will introduce this final point and the subsequent chapter will delve more deeply into it.

[124] Resource constraints and data collection options normally influence the choice of valuation techniques. See Barbier et al., above n. 71, at 40.

7. Did the well run dry or is there another novel in there? (see chapter III)

This chapter investigated the efficiency of methods of environmental damage assessment considering that the traditional remedy in tort law was represented by the obligation of the polluter to provide monetary compensation equal to the harm that had been caused. However, there is a tendency in the law to move away from the (mere) imposition of damages and to impose restoration of the damaged environment as primary remedy. While a number of advantages seems to justify such evolution, there are also several limitations that need to be investigated. The next chapter will use the economic approach to compare advantages and disadvantages of restoration as opposed to the monetary compensation. Moreover, the economic approach will help to provide a model on how remedies in specific cases of environmental harm should be structured in order to attain not only restoration in a cost-effective manner but also optimal deterrence.

Take-aways from chapter II and bridge to chapter III

- There is ‘no one size fits all’ method of environmental damage assessment in the environmental economics domain.
- Three dimensions may be considered in order to estimate the efficiency of conventional methods in view of optimal deterrence. These are: the accuracy, the captured value of nature and the assessment costs.
- In the event of more substantial losses in use values, market-based and revealed preference approaches can achieve more optimal deterrence.
- In the event of more substantial losses of non-use values (or unique values), stated preference approaches can achieve better deterrence provided that the non-use component of value is sufficiently high to outweigh the higher assessment costs and that very scrupulous procedures of assessment are conducted.
- The most efficient method shall be determined based, on the one hand, on the specificities of the injured environment and, on the other hand, on the data and resources available.

PART II

Are remedies in legislation and case law inducing optimal deterrence and cost-effective restoration?

CHAPTER V

Comparing the Efficiency of Remedies at the National Level: US v. EU*

This chapter continues the previous analysis of environmental liability laws, moving from the international to the national level. It provides a detailed picture of the law on environmental damage assessment in the US and the EU. Of particular interest is the evolution of US natural resource damage assessment law, which shows how the focus shifted from developing a suitable method for monetarily quantifying the total economic value of nature to developing cost-effective restoration plans that provide equivalent services. As Jones and DiPinto (2018) observe, the 'restoration-based' compensation has become the primary remedy and ‘compensatory restoration’ is specifically directed towards interim losses and irreversible damage. During the 1990s, the results of this long evolution were transplanted to the EU, leading to the adoption of the EU Directive on Environmental Liability (ELD) in 2004. This proposed to remedy (and prevent) environmental damage with a set of restoration actions modelled after US laws. However, the ELD provided less detailed guidance than the US law on natural resource damage assessment. Furthermore, a number of obstacles may prevent the achievement of full restoration even when the damage is reversible. These include information costs for polluters, a lack of guidelines on primary and compensatory restoration, the impossibility of identifying liable parties and a lack of time constraints in litigation. Lastly, technical uncertainties in the EU are more pronounced due to the lack of scholarship in environmental economics and the absence of experts capable of conducting accurate equivalency analyses compared with the US. In conclusion, there are several reasons to doubt that the obligation to restore introduced by the ELD can result in polluters being held fully liable for the full costs of restoration and that it can induce optimal deterrence and cost-effective restoration.

1. US Law

If one wishes to identify the onset of laws allowing claims for environmental damage in the US, there seem to be two possible beginnings.

Based on the most widespread view, the starting point of laws and techniques for Natural Resource Damage Assessment (hereinafter, NRDA) is represented by the Exxon Valdez oil spill of 24 March 1989. Indeed, after the Exxon vessel ran aground on Prince William Sound (Alaska) and the pictures of dying birds and sea otters featured on prime-time news across the globe, hundreds of experts ranging from marine biologists and ecologists to economists and oil spill trajectory modelers were appointed by federal, state and local governments to help with litigation.[1] Experts were also hired from the other side by Exxon and they all tried to study the effects of the injury and to put a dollar value on it. While chapter VI will provide an in-depth analysis on the valuation of the damage in this and other oil pollution cases, the focus here is on the content (and the evolution) of liability laws.

After the accident, the US Congress approved the Oil Pollution Act (OPA) of 1990 primarily to improve the legal response to oil spills[2] and make it more coherent and safer.[3] In addition to that, the new law wanted to provide governments and private parties with more effective tools to recover natural resource damages for losses caused by oil spills to public and private interests. Based on that, private and public parties were entitled to claim economic and environmental damage from liable parties (e.g., the spiller) or the federal trust fund.[4]

The second and less well-known account is that the history of federal statutes allowing claims for environmental damage started in the 1970s,[5] well before the approval of the OPA in 1990[6] and, more precisely, in 1973, when the Congress adopted the Trans-Alaska Pipeline Authorization Act (TAPAA) to address the environmental effects of oil spills possibly caused by

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* An early version of this chapter has been published with the title 'Comparing Remedies for Environmental Damage: US v. EU' in the Comparative Law Review Special Issue 13/1 'Rescuing Comparative Law And Economics? Exploring Successes And Failures Of An Interdisciplinary Experiment' (2022), upon invitation of the editor, Prof. Bellantuono.

[1] V. Ann Lee, P.J. Bridgen & Environmental International Ltd. (EI), *The Natural Resource Damage Assessment Deskbook: A Legal and Technical Analysis*, Environmental Law Institute, Washington, at 7 (2002).

[2] The exact words of the Senate Report helps understand the precise goal of OPA: "what the Nation needs is a package of complementary international, national, and state laws that will adequately compensate victims of oil spills, provide quick, efficient clean-up, minimize damages to fisheries, wildlife and other natural resources and internalise those costs within the oil industry and its transportation section" (S. Rep. No. 101-94 at 2 (1989) reprinted in 1990 U.S.C.C.A.N. 722, 723).

[3] J. Chapman, 'From Bligh Reef to the Gas Pump', 15 *The Mesquite Review* 45 (2000), cited in Lee and Bridgen, above n. 1, at 7, footnote 2.

[4] It is worth noting that the American Congress addressed the new accident by looking at the previous failing experience of the 1978 Amoco Cadiz spill in France. Indeed, the Amoco Cadiz represented the previous and biggest oil spill before the Exxon Valdez and in 1989 the related lawsuits were still ongoing, but no damages had been paid. See next chapter VI.

[5] The decade started with the Clean Air Act (CAA) in 1970 and, then, the Federal Water Pollution Control Act (FWPCA) in 1972, two command-and-control regulatory tools that worked through penalties for infringements. However, the Congress soon acknowledged that penalties were not sufficient to achieve environmental restoration in the aftermath of incidents. In order to overcome the limitations of penalties, as well as of the common law damage theory relying on monetary compensation, the Congress turned to a liability regime with an innovative approach to damages. The new approach was indeed based on restoration rather than market valuation to ensure that recovered sums were spent to return the environment back to baseline conditions and, thus, to make the public whole. See Lee and Bridgen, above n. 1, at xv.

[6] The origins of the NRDA seem to trace back to 1962, when the famous environmental science book by Rachel Carson, *Silent Spring,* was published. It unveiled the negative consequences of DDT and other toxic chemicals, hence triggering the explosion of the modern environmental activism, with a sound impact on the public opinion, as well as son policy-makers.

tankers transporting oil from the terminal of the pipeline to continental US cities. The TAPAA indeed made the pipeline's holder strictly liable to all damaged (public or private) parties and regardless of the public/private ownership of damaged natural resources. Moreover, vessels' operators were made strictly and jointly liable for all damages, including clean-up costs, born by any public or private party as a result of the accident. In addition to immediate response action and clean-up, the TAPAA for the first time allowed recovery for damages to the environment, the so-called 'natural resources damages', even if it did not explain how to measure them or whether damages to natural resources not linked to human uses could also be recovered.[7] The compensatory principle inspiring the TAPAA was later expanded by the Deepwater Port Act (DPA) of 1974. The DPA imposed liability for human and environmental damage caused by an oil spill at a deepwater port.[8] Moreover, it identified the Secretary of Transportation as trustee of the marine natural resources entitled to recover the 'sums that may be necessary to Federal and State governments to restore fisheries, the habitats of sedentary living species or to replace estuarine areas or other coastal resources damaged by oil or natural gas*'*.[9] However, like the TPAA, the DPA did not provide any guidelines on how damages should be measured. It was later incorporated in the OPA and repealed by it in 1990. Until that moment, it remained the reference point for natural resource damage liability.[10]

In 1977, the Amendments to the Clean Water Act (CWA) expanded the scope of the DPA by empowering the official representatives of any States with the right to act on behalf of the public (as trustees) and to recover the costs of restoring and replacing injured, damaged or destroyed natural resources into US waters (and contiguous zones).[11] Then, the sums had to be used to restore or acquire equivalent resources by the 'appropriate agencies of the Federal Government, or the State Government'.[12] Thanks to the amended CWA, many federal authorities were involved in oil spill responses in the 1980s and several scientists and economists started to develop techniques of environmental damage assessment that ultimately led to the NRDA as we know it today .[13]

[7] Lee and Bridgen, above n. 1, at 10.
[8] Yet, the applicability of the DPA was very limited given that only one deepwater port was licensed at that time, *i.e.* the Louisiana Offshore Oil Port (Lee and Bridgen above n. 1, at 11, footnote 19).
[9] 1974 U.S.C.C.A.N. 7529, 7544.
[10] Lee and Bridgen, above n. 1, at 12.
[11] 33 U.S.C. § 1321(b)(3), (f), ELR Stat. FWPCA §311(b)(3), (f).
[12] 33 U.S.C. § 1321(f)(5), (f), ELR Stat. FWPCA §311(f)(5).
[13] For an overview of significant cases from 1967 to 1991, see NOAA (National Oceanic and Atmospheric Administration), Herbert Charles Curl, Kenneth Barton and Lori Harris, "Oil spill case histories, 1967-1991: summaries of significant US and international spills", 1992, Report (United States. NOAA. Hazardous Materials Response and Assessment Division) no. HMRAD 92-11 URL : https://repository.library.noaa.gov/view/noaa/1671 [accessed 20 November 2023]

The passage of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) in 1980 was another major milestone in the history of the US environmental damage assessment. Although oil was excluded from the list of hazardous substances, the statute extended liability for environmental injuries to all natural resources rather than just waters. Admissible damages included the costs of restoring or acquiring alternative habitats and the costs of assessment. However, the number of NRDA claims significantly increased after the 1986 Superfund Amendments.[14]

In 1988, the Amendments to the Marine Protection, Research and Sanctuaries Act (MPRSA) introduced liability for any losses or injuries to sanctuary resources.[15] For the first time, liability for environmental damage was detached by a specific cause (e.g., oil spills or releases of hazardous substances) and damages associated with the 'lost use' of the resource during the time of the injury (so-called, interim losses) were made explicitly recoverable.[16] To sum up, the new MPRSA allowed trustees to recover restoration costs, interim lost uses and assessment costs. The 1990 National Park System Resources Protection Act (NPSRA) was modelled after the MPRSA, covering injuries to park resources independently from the cause and admitting the same categories of damages.[17]

It Is clear from the above that the approval of OPA in 1990 has to be interpreted as a crucial but subsequent step within the history of NRDA. Built on the model of the 1974 DPA, OPA provided a comprehensive liability scheme for natural resource damages caused by oil spills. It expanded the scope of the previous legislation (CERCLA) by adding the right to recover private party damages caused by oil spills and, thus, aiming at making whole not only the public but also individuals. The metric for damages was always given by restoration costs rather than the market value.

It can be concluded from the above that it would be too simplistic to believe that NRDA began in the 1980s. Indeed, the assessment of natural resources was already taking place in the US in the '60s and '70s. However, only in the late '80s 'NRDA came into full swing as a legal and scientific discipline'.[18]

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[14] The Amendments introduced rebuttable presumptions in favour of the plaintiffs and included Indian tribes as authorized trustees.

[15] 16 U.S.C. § 1443(a).

[16] 16 U.S.C. § 1432(6).

[17] 16 U.S.C. § 19jj-1(a).

[18] Lee and Bridgen, above n. 1, at 16.

1.1 NRDA terminology and choice of the law

Considering that federal statutes on NRDA share a common legislative history, it is worth bearing in mind the meaning of the terms which are common to all federal laws:

- 'injury' is the scientific concept that refers to the adverse impact on natural resources as a result of an incident (i.e., oil spill or release of hazardous substances);
- 'natural resources' are 'land, fish, wildlife, biota, air, water, ground water, drinking water, and other resources belonging to, managed by or held in trust by, or otherwise controlled by the United States, any State or local government, any foreign government, any Indian tribe';[19]
- 'damage' is the legal concept or the translation of the injury into what the liable party has to do or to pay to compensate accidents' victims;
- 'restoration' is the activity aiming at returning the injured natural resources to the conditions that would have existed but for the incident (baseline);
- 'trustees' are the federal,[20] state or tribal public bodies whose role is to protect the public interest and to 'make the public whole' in case of injuries affecting resources under their trusteeship (trustee's management, control and ownership). Trustees can bring claims and, if injuries harm resources under the trusteeship of multiple agencies, NRDA is conducted by intergovernmental teams of technical experts representing all interested agencies, further supervised by the respective attorneys.
- 'natural resource damage authority' is the response and clean-up authority, for instance the US Environmental Protection Agency (EPA) is the federal authority with primary responsibility under CERCLA, but other trustees assess natural resource damages.

Even if legal principles are common to all statutes, the consequences in terms of exact NRDA activities to implement and the timeframe of action varies according to the applicable statute. For this reason, after the accident, it is essential to immediately figure out the geographic location of the incident (water, land, national park or marine sanctuary) and the agent causing the injury (oil, hazardous substance or any physical trauma). Then, it is possible to establish the applicable law. The table on the next page outlines this match.

[19] 42 U.S.C. § 9601(16), ELR Stat. CERCLA § 101(16). The definition is exactly the same in OPA § 1001(20).
[20] For an overview of designated federal trustees, see Lee and Bridgen, above n. 1, at 157ss.

	US Statutes Laws				
	CERCLA	OPA	CWA	NPSRA	MPRSA
Cause injury	Hazardous substance	Oil	Oil and hazardous substances	Any	Any
Location	Any	Navigable US waters and EEZ[21]	Navigable US waters and contiguous zones	Within a park	Within a marine sanctuary

Table 1 [US statute laws on pollution][22]

All US statute laws on pollution provide a right of legal action to government agencies with the management, ownership or control on the injured resources. The trustee entitled with this right depends on what kind of resource has been affected (marine mammals, terrestrial mammals, fish, threatened, endangered or protected species, habitat, land, groundwater).[23]

Lastly, the timeframe of action depends on the characteristics of the injury (whether it's acute or chronic). Trustees have to respond quickly by collecting evidence and contacting potential polluters in case of acute events (e.g., oil spills in water and hazardous waste) that might provoke serious impacts on wildlife.[24] In addition, trustees can collect evidence thanks to the availability of a fund created by OPA.[25] Lastly, improved coordination among all involved public bodies is likely to speed up the NRDA and lay the foundations for early settlement. These adjustments allowed the trustees to be better placed to respond to environmental injuries and promptly mitigate their effects. That, in turn, contains the magnitude of the injury and reduces the liability of polluters.

[21] The EEZ are the Exclusive Economic Zones. They were added after the Exxon Valdez case.

[22] Source: Lee and Bridgen, above n. 1, at 22.

[23] See 40 C.F.R. § 300.600 for the precise match between public agencies, management areas and trustee resources.

[24] Providing immediate response was nearly impossible in the early versions of the NRDA. Only after the Exxon Valdez, response teams have been set up by public agencies and the industry.

[25] See 33 U.S.C. § 2713, ELR Stat. OPA § 1013 that created a $1 billion Oil Spill Liability Trust Fund that absorbed the previous Fund under the Deepwater Port Liability Act and the Trans-Alaska Pipeline Fund. The reader should be aware that a fund (Superfund) was also created to ensure a swift response of the government under CERCLA. Yet, its use by the trustees has been precluded in 1986 (Superfund Amendments), whereas the OPA fund is available to trustees for uncompensated removal costs and damages that have not been directly paid by liable parties or where there is no identified responsible party (Lee and Bridgen above n.1, at 111). The Fund is filled with taxes (including an oil tax from the industry), cost recoveries from responsible people for oil spills and civil penalties always incurred by liable parties for oil spills. Compensable costs under the Fund include removal actions and activities to initiate NRDA, as well as to conduct restoration, to cover personnel costs, implementation and enforcement of OPA, research and development (OPA §1012(a)).

1.2 Response actions

Within the NRDA, a preliminary distinction needs to be made between clean-up and restoration. Under section 104 of CERCLA, EPA has the power to adopt so-called 'response actions'[26] in the immediate aftermath of a release of hazardous substances. Response actions serve to assess, control and clean up sources of contamination to protect human health and the environment. They include removal and remedial actions with no clear dividing line.[27]NRDA activities are distinct from response actions as the latter aim at restoring or compensating past or ongoing damage (residual to clean-up).

'Removal actions' are implemented (by EPA) either to respond quickly to an emergency or to carry out clean-up in a short period of time.[28] If they draw on Superfund, they need to last no more than 12 months and not require more than $2 million.[29] Apart from this, they are subject to less prescriptive requirements for clean-up levels and alternatives analysis.

'Remedial actions', instead, take place at complex sites and they require much more time,[30] since their aim is to achieve a permanent remedy for site contamination. They are subject to much more stringent requirements when it comes to clean-up standards and the selection of action to implement. The coordination between response actions and NRDA is crucial to make the whole process more cost-effective and to make polluters pay less for residual damage under NRDA.[31] Also, US laws favour settlements and voluntary clean-ups.[32]

Whether clean-up is ongoing or it has ended,[33] natural resource damage provisions wish to restore the injured resources to the baseline, hence addressing damage residual to clean-up or even past historical damage. Indeed, 'natural resource damages are viewed as the difference between the natural resource in its pristine condition and the natural resource after the clean-up, together with the lost use value and the costs of assessment. As a residue of the clean-up action, in effect damages are thus not generally settled prior to a clean-up settlement.'[34]

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[26] See 42 U.S.C. §9601(25), ELR Stat. CERCLA § 101(25).

[27] J. L. Anderson, 'Removal or remedial?: The Myth of CERCLA's two response system', 18 *Columbia Journal of Environmental Law* 103 (1993), cited in Lee and Bridgen, above n.1, at 38, footnote 35.

[28] See 40 C.F.R. § 300.415.

[29] See 40 C.F.R. § 300.415(b)(5).

[30] See 42 U.S.C. § 9604, ELR Stat CERCLA § 104.

[31] "Sometimes it is better for responsible parties to pay more in clean-up expenses to avoid a big natural resource damage that outstrips the incremental cost of the additional clean-up" (Lee and Bridgen, above n.1, at 40). This will come back in the analysis of the Exxon Valdez, see chapter VI.

[32] See Section 122 of CERCLA.

[33] See 42 U.S.C. § 9607(a)(4)(C), (f), ELR Stat. CERCLA § 107(a)(4)(C), (f). There is no requirement under CERCLA for which clean-up has to be completed before a NRD claim can be brought.

[34] The citation comes from the US District Court for the District of Massachusetts in a 1989 legal proceeding. For the full quote, see Lee and Bridgen, above n.1, at 46, footnote 61.

1.3 Recoverable costs

Under section 107 of CERCLA, it is possible to recover damages for injury, destruction or loss of natural resources, including the reasonable costs of assessment.[35] Moreover, the public agency is obliged to use the recovered sums to restore, replace or acquire the equivalent of the harmed natural resources, although recovered sums 'shall not be limited by the sums which can be used to restore or replace'.[36] Likewise, OPA sets down that those responsible for an actual or threatened discharge of oil are liable for both 'removal costs' and damages.[37]

'Removal costs' are broadly defined under OPA as the costs incurred after a discharge or threatened discharge to 'prevent, minimize, or mitigate oil pollution from an incident'.[38] In addition, removal has been defined as the containment and removal of oil from water and shorelines, plus other actions that may be 'necessary to minimize or mitigate damage to the public health or welfare including, but not limited to, fish, shellfish, wildlife, and public and private property, shorelines and beaches'.[39] Given this broad definition, the exact identification of what is a removal cost depends on the discretion of those who judge on cost-recovery claims.[40]

Claimants can be either public entities or private parties incurring response costs and damages.[41] They are granted election of remedies, meaning that they can choose between filing the fund or responsible parties. Whatever they choose, however, before bringing an action, claimants shall first demand responsible parties (or guarantors) for removal costs (and damages).[42] Only in case of a denial of liability or a failure to settle within 90 days, claimants can present their claims.[43]

In conclusion, recoverable damages under OPA are much broader than under CERCLA, CWA, MPRSA. Liable parties have to respond to six categories of damage, plus the costs to assess them:

i) damage to real and personal property (economic losses from the destruction or injury of real and personal property);[44]

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[35] See 42 U.S.C. § 9607(a)(4)(C), ELR Stat. CERCLA § 107(a)(4)(C).
[36] See 42 U.S.C. § 9607(f)(1), ELR Stat. CERCLA § 107(f)(1).
[37] OPA also amended several related laws in order to strengthen civil and criminal penalties, e.g. OPA § 4301-4303 amended some sections of the CWA.
[38] OPA § 1001(31).
[39] OPA § 1001(30).
[40] For instance, it seems that the interpretation given by the Coast Guard administering the Fund under OPA tends to give a very narrow interpretations of what should be included in the category of removal costs. Not all that private parties think is a removal action is indeed compensable by the Fund (Lee and Bridgen, above n.1, at 101, footnote 72).
[41] OPA § 1001(4), 1002(b). However, private parties have a heavier burden of proof compared to public entities: they have to prove that their acts are in line with the National Contingency Plan (OPA § 1002(b)(1)(B). CERCLA, like OPA, creates a similar two-tiered framework for cost-recovery actions, as public entities can recover also costs which are not consistent with the National Contingency Plan (CERCLA §107(a)).
[42] OPA § 1013(a).
[43] OPA § 1013(c). Claims are indeed dismissed until they comply with OPA's pre-suit provisions.
[44] OPA § 1002 (b)(2)(B).

ii) loss of profits or earning capacity as a consequence of the injury, destruction or loss of real property or natural resources;[45]
iii) loss of subsistence use from the injury or destruction of natural resources;[46]
iv) loss of taxes and public revenues (royalties, rents, fees);[47]
v) damage for increased costs of public services, such as safety activities and any other service to protect against hazards caused by oil spills;[48]
vi) natural resource damage (can be claimed only[49] by designated trustees, states, Indian tribes or foreign governments,[50] provided that they 'develop and implement' a plan for the 'restoration, rehabilitation, replacement or the acquisition of the equivalent of the injured resources under their trusteeships'[51] and sums recovered by trustees are invested in restoration plans and assessment costs);[52] natural resource damage shall be measured as including: the costs of restoring, rehabilitating, reacquiring the equivalent of the damaged resources, their diminution in value pending restoration and the reasonable costs of assessing them.[53]

1.4 The process of damage assessment

Notoriously, the main goal of the NRDA is to allow trustees to recover the costs and damages illustrated above, provided that they can prove the nature and extent of injuries caused by the accident to resources under their trusteeship, that they manage to convert the injury into a monetary sum or a set of activities to compensate the public for the injury and, lastly, that they provide evidence that the response actions (see above) are not sufficient to adequately remedy the injury without additional measures.[54] If claims are successful, trustees are obliged to employ

[45] OPA § 1002 (b)(2)(E).
[46] OPA § 1002 (b)(2)(C). This category of damages is particularly helpful for private parties and members of tribes.
[47] OPA § 1002 (b)(2)(D). This head of damage is clearly at the disposal of the US, a state or a political subentity.
[48] OPA § 1002 (b)(2)(F).
[49] Citizens are excluded from standing for natural resource damage claims, but OPA § 1006(g) allows any person to sue a federal official in case of alleged failure to perform a mandatory duty under § 1006. In addition, Courts may award prevailing parties litigation costs including attorney and expert witness fees. This citizen suit provision is peculiar to OPA, unlike other federal statutes on NRDA. This means that trustees' duties concerning NRDA can be enforced by private parties who are particularly incentivised to do so thanks to reduced litigation costs.
[50] OPA § 1002 (b)(2)(A).
[51] OPA § 106(c)(1)-(4).
[52] OPA § 1006 (f).
[53] OPA § 1006(d)(1).
[54] 43 C.F.R. § 11.23.

recovered costs and natural resource damages for restoration. The steps of NRDA are the same under CERCLA/OPA:[55]

- Preassessment phase (ending in a Preassessment Screen Determination)

Trustees carry out an initial screen to value whether or not a natural resource damage assessment is needed, based on their jurisdiction and the information available about the injury.[56] In emergency situations, trustees request immediate response action to the National Response Center. Lacking sufficient response, they can carry out immediate response action to avoid further damage and implement off-site activities to avoid that the polluting substance migrates and affects other resources within their trusteeship.[57]

- Assessment phase (ending in an Assessment Plan)

This phase serves to prove that trustees developed an approach to assessment ‘of reasonable cost’ and that responsible parties have been invited to participate.[58] With the plan, trustees can choose between the type A and/or type B assessment procedures.
Type A procedures are simplified, fast and cheap,[59] but limited to environmental damage up to $100,000[60] and in special areas (Great Lakes and coastal/marine environments).[61]
Type B procedures involve extensive data collection and analysis, plus a preliminary estimate of damages.[62] This estimate serves to ensure that the assessment costs are appropriate given natural recovery and the expected returns of litigation.[63]
After the choice between Type A and Type B has been taken, the real assessment can start. It consists of: 1) injury determination; 2) injury quantification; 3) damage determination.

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[55] They are more simplified under OPA. For instance, restoration projects are used as a metric for valuation of both components of damage, compensation to return the environment to the baseline and also interim losses. So, interim losses, whereas under CERCLA regulations interim losses are estimated based on their monetized value. For a detailed overview see Lee and Bridgen, above n.1, at 194ss.
[56] Indeed, one of the criteria to justify a NRDA is the availability of data for a damage assessment at a reasonable cost (43 C.F.R. § 11.23).
[57] It is the trustee’s burden to prove in NRDA claims that only emergency activities were conducted and they were all reasonable and necessary.
[58] The regulations require that trustees make reasonable efforts to identify the responsible parties (43 C.F.R. § 11.32(a)(2)). Once they have been determined, the assessment plan is notified to them and they are invited to participate to the plan development. This procedure aims indeed at ensuring protection to responsible parties.
[59] They require minimal field observation and they rely on computer models (the NRDA Model for Great Lakes and the NRDA Model for Coastal/Marine Environments). Data needed include: characteristics of the released substance (type and volume), characteristics of the release (time, duration, location), tide and wind conditions and the extent of response actions. Some parameters of the model concerning air and water temperature or habitat type can be changed by the official working on the assessment, if more accurate data are available. (Lee and Bridgen, above n.1, at 192).
[60] However, they tend to be preferred in collaborative assessments by both trustees and responsible parties.
[61] 43 C.F.R. § 11.42.
[62] If data are available, otherwise the estimate is required only after the injury determination.
[63] 43 C.F.R. § 11.38.

The purpose of the 'injury determination' step is to prove that an injury (a measurable adverse change in the chemical or physical quality of a resource) occurred, by means of 'acceptance criteria' defined by the regulations for each category of resources[64] and 'pathways of exposure' to prove causation.[65]

In the following 'injury quantification' step, trustees estimate the scope of the injury by measuring the diminution in services from pre-incident conditions.[66] These are the conditions that would have existed but for the accident (baseline). They can be obtained by looking at historical data or by comparing a control area with the affected one (when historical data are not available).[67] The resource recoverability is considered, i.e. the time needed to recover with and without restoration actions.[68]

With the last 'damage determination' step, trustees convert the injury into amounts of money or sets of activities aimed at compensating the public for natural resource injuries by returning resources/services to the baseline. The choice of activities requires to consider first 'natural recovery alternatives',[69] then 'active primary restoration actions' (aimed at directly returning the environment to the baseline[70] 'on an accelerated time frame'[71]) and, lastly, 'compensatory restoration actions' that 'provide services of the same type and quality, and of comparable value as those injured'.[72] The corresponding costs can be then recovered:

1. Costs to restore, rehabilitate, replace or acquire the equivalent of resources lost. For these, trustees need to develop a Restoration Plan by identifying a 'reasonable range of restoration alternatives' scaled to the injury and their costs. The reasonability of alternatives depend on technical feasibility and cost-effectiveness.
2. Compensable value of services pending recovery to baseline[73] (interim lost uses). This value is allowed only for losses involving 'committed uses', i.e. current or planned public

[64] 43 C.F.R. § 11.64.
[65] 43 C.F.R. § 11.63.
[66] 43 C.F.R. § 11.71(a).
[67] 43 C.F.R. § 11.71(c), (d). More details on methods for determining the baseline can be found in Lee and Bridgen, above n.1, at 244ss. Broadly speaking, baseline conditions can be determined through historical data, reference sites, reference populations and gradient (stressors) designs.
[68] 43 C.F.R. § 11.73.
[69] This means that 'no human intervention would be taken to directly restore injured natural resources and services to baseline' (15 CFR § 990.53 – Restoration selection).
[70] The baseline is defined as "the condition of the natural resource that would have existed had the incident not occurred. Baseline data may be estimated using historical data, reference data, control data, or data on incremental changes (e.g., number of dead animals), alone or in combination, as appropriate" (15 CFR § 990.53).
[71] 15 CFR § 990.53.
[72] Ibid.
[73] 43 C.F.R. § 11.80.

uses of natural resources based on legal, administrative or financial commitments taken before the accident.[74]

3. Reasonable costs of assessment, including monitoring, oversight costs and enforcement costs.[75]

- Post-assessment phase

Once the Assessment Plan is completed, trustees are requested to demand responsible parties to pay for the recoverable costs. A period of 60 days is allowed for negotiation before filing the suit.[76] The sums recovered then must be deposited in a special US Treasury account for NRDA and used to effectively restore the injured environment. Multiple trustees involved in NRDA usually agree on forming trustee councils to implement restoration actions.

1.5 The damage determination and the interpretation of the regulations

The phase of 'damage determination' within the second assessment phase is about putting a dollar value on harmed natural resources in order to recover the costs of restoration and, in this way, compensating the public and the environment for the injury.[77]

None of the mentioned federal laws set down precise techniques of damage estimation. The US Congress requested that the two main agencies (DOI and NOAA) issued regulations with the 'best available' techniques. That required at least two decades of academic discussions, public debates and fierce litigation.[78] CERCLA regulations were released in 1987 and OPA regulations in 1996. They both were challenged in litigation and the case-law helped develop a robust knowledge on techniques of NRDA in the US. Yet, controversies over methods never really ended.

[74] The "committed use approach" to interim losses was upheld by the Court in Ohio v. Department of the Interior, 880 F.2d 432, 462, 19 ELR 21099, 21115 (D.C. Cir. 1989).

[75] 15 C.F.R. § 990.30. US governments conducting NRDA generally make a distinction between direct and indirect costs of assessment. The former refers to the expenses for a single environmental damage case, whereas indirect costs are those with multiple objectives, such as fringe benefits. Although indirect costs may be considerable, US Courts are expected to allow them, given that 'allocating indirect costs that cannot be directly accounted for as costs of a specific project is a well-established accounting practice' (Kennecott Utah Copper Corp. v. Department of the Interior, 88 F.3d 1191, 1223-1224, 26 ELR 21489, 21501-02 (D.C. Cir. 1996).

[76] 43 C.F.R. § 11.92.

[77] The reader should be aware that converting the value of natural resources into an amount of money has always been quite controversial. Some people raise objections because natural resources are unique, not substitutable and merely priceless (Leopold and the conservationists). Others are in favour of pricing nature since in a market-based economy there cannot be adequate protection and conservation of the environment without monetary valuation.

[78] Lee and Bridgen, above n.1, at 282.

The discussion below summarises the main issues of controversy that triggered the American debate on methods of environmental damage assessment and the final approach that currently informs the interpretation of CERCLA and OPA regulations.

The most debated issue was whether environmental damage estimates should be based on values or restoration costs and, if on values, whether these should be limited to market prices or they should include the 'total value' (use and non-use values) that the society places on injured natural resources.[79]

As to the cost versus value issue, the first set of regulations issued by the DOI originally relied on market values for the assessment of restoration, meaning that restoration projects were allowed provided that their costs did not exceed the market value of the injured resources. This point was challenged in *Ohio* v. *Department of the Interior*, [80] since most of the times restoration projects go beyond market values.[81] With this case, Ohio and other States challenged the new regulations issued by the US Department of Interior (DOI) to specify the techniques for the assessment of environmental damage under CERCLA. The issue at stake regarded:

- the fact that damages had to be limited to "the lesser of the costs" of restoration or the lost use value under the NRD assessment regulations;
- the hierarchy of techniques which gave priority to market-based techniques over nonmarket valuation techniques;
- the inclusion of CV as a possible technique adding that 'estimation of option and existence values (i.e., non-use values) shall be used only if…no use values can be determined'.[82]

With its decision, the Court of Appeals for the District of Columbia stated three main principles: First, the main purpose of NRD should be to restore the damaged environment and, for this reason, damages should be based on restoration costs (the cost of a restoration project) rather than use values (unless 'grossly disproportionate to use values').[83]

Secondly, judges should be always allowed to compensate for non-use values (it would be unreasonable to give only priority to use values and not to include non-use values). After defining

[79] Which are beyond the uses and services that can be derived from natural resources and traded in the marketplace. Nonmarket techniques have been therefore put in place to estimate the non-use component of the value of natural resources, e.g. the existence value, for which people's willingness to pay is not captured by markets.
[80] 880 F.2d 432 (D.C. Cir. 1989).
[81] The decision of the D.C. Circuit was then incorporated with the 1994 Type B rule, which was again challenged in *Kennecott Utah Copper Corat v. Department of Interior* (1996).
[82] 43 CFR § 11.83(b)(2).
[83] In other words, the D.C. Circuit held that the 'lesser of' rule was invalid since it was in contrast with the intentions of the Congress. Based on this rule, damages could be recovered as the lesser of restoration/replacement costs or the diminution of use values (43 C.F.R. § 11.35(b)(2)). By contrast, the Parliament clearly expressed preference for restoration costs as a measure of recovery (880 F.2d 432 D.C. Cir. 1989, par. 459).

passive-use (or non-use) values, the Court stated that they should be, *prima facie*, included in the damage assessment. Excluding non-use values was inconsistent with federal laws and trustees had to be allowed to recover them.[84] However, the preamble of the 1994 regulations announced that standards for the estimation of non-use values would have been introduced by the DOI and that never happened. Nor *Ohio* ruled on that point. Therefore, the gap in the US legal framework still remains and it represents an issue only for the calculation of interim losses, since it can lead to very high estimates.[85]

Thirdly, nonmarket valuation techniques (CV) should be used as much as market-based techniques (giving priority to market-based valuation and appraisal techniques would be unreasonable).[86] The Court did not endorse the industry's arguments for which the method was imprecise, biased and likely to induce overestimation.[87] It supported the use of CV as the only economic technique by which the total value of natural resources could be measured by economists.

In *Kennecott Utah Copper Corp. v. Department of Interior* (1996) the Court confirmed its previous position in *Ohio* and affirmed that trustees were allowed to employ the controversial contingent valuation technique to estimate passive use values since it can produce useful and reliable results.[88] It was also cleared out that the cost-effectiveness of restoration alternatives should not be a mandatory criterion for selection and that the not gross disproportionality criterion remains applicable when comparing and selecting restoration options.[89]

Lastly, on assessment costs, the D.C. Circuit upheld the regulations, confirming that reasonable assessment costs, recoverable under the regulations, could include monitoring and oversight costs, but not attorney fees.[90]

[84] On this point, see F.B. Cross, 'Natural Resource Damage Valuation', 42 *Vanderbilt Law Review* 269 (1989), saying that the option and existence value of national parks is quite large based on surveys' results.

[85] In this respect, a distinction between CERCLA and OPA regulations must be considered: the former ask to recover the economic value while the latter focus on the cost of restoration options that can provide service and/or resources equivalent to the lost ones (compensatory restoration). So, interim losses can be recovered under OPA only when alternative restoration options are not available.

[86] *Ibid,* par. 463.

[87] Ohio, 88 F.3d at 476, 19 ELR at 21123-24.

[88] General Elec. Co., 128 F.3d § 772-74, 28 ELR § 20265-66. The D.C. Circuit therefore confirmed its previous position expressed in the case with the Department of Interior and concerning NRDA regulations under CERCLA.

[89] So, under CERCLA regulations trustees are not obliged to select the most cost-effective restoration option.

[90] OPA regulations were challenged under this point in *General Electric Co. v. Department of Commerce* by a number of chemical, oil, and insurance industries, with thirteen states and the National Resource Defence Council assisting the NOAA to defend the regulations.

1.6 The US practice: from contingent valuation to restoration (and beyond)

In order to understand the development of the US case law on natural resource damage assessment, a previous clarification needs to be made. While the above-mentioned laws were approved (especially, CERCLA in 1980), environmental economists were conducting research on how to value the environment. Particularly, in the late 1980s they had already developed both market-valuation techniques and non-market valuation techniques. The latter aimed at assessing the value of non-market goods (environmental goods) which, in spite of the absence of market prices, have nevertheless value because of their direct use (use values) or their mere existence (non-use values). Especially the contingent valuation technique was receiving much attention at that time because it seemed to be the only way to calculate the non-use value and to get closer to the total value of the environment.[91] The first landmark case in the US came therefore in the midst of the new adopted laws on NRDA and the developments in the field of environmental economics.

Just four months after the Exxon Valdez oil spill, the cited *Ohio v. DOI* came in the spotlight to trigger the already lively debate on the valuation of nature. The ruling was extremely relevant because it overturned the regulation by putting on the same level of importance both restoration and contingent valuation. In this way, the Court wanted to overcome the previous trend of calculating environmental damages by only looking at market prices and it opened the road towards the calculation of non-use values through the CV method. After the Ohio Court expressed its favor for the CV, it was applied in the Exxon Valdez case leading to a final amount of damages around US$ 9 billion.[92] Likewise, in the case *United States v. Montrose Chemical Corp.* in Southern California, damages for environmental damage were awarded for over half of US$ 1 billion. Moreover, these decisions triggered considerable debate among legal scholars around restoration costs versus lost use values.[93] Scholars were split between those supporting the use of CV (Montesinos, Dobbins, Brookshire & McKee, McConnell, Baker),[94] those limiting

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[91] See chapter II for a deeper discussion on this point.

[92] R.T. Carson, R.T. Mitchell, M. Hanemann, R.J. Kopp, S. Presser & P.A. Ruud, 'Contingent Valuation and Lost Passive Use: Damages from the Exxon Valdez', 25 *Environmental and Resource Economics* 257 (2003). However, the case was settled for US$ 1 billion in the end, plus fines, compensation and clean-up costs, plus a lawsuit for punitive damages that were reduced to $500 million in 2008 by the Supreme Court. See chapter VI.

[93] For a summary of the whole debate between 1989 and the late 1990s, see D. B. Thompson, 'Valuing The Environment: Courts' Struggles With Natural Resources Damages', 32(1) *Environmental Law* 57, at 62 (2002).

[94] M. Montesinos, 'It May Be Silly, but It's an Answer: The Need to Accept Contingent Valuation Methodology in Natural Resource Damage Assessments', 26 *Ecology Law Quarterly* 48 (1999); J.C. Dobbins, 'Pain and Suffering of Environmental Loss: Using Contingent Valuation to Estimate Nonuse Damages', 43(4) *Duke Law Journal* 879 (1994); D. Brookshire & M. McKee,'Is the Glass Half Empty, Is the Glass Half Full? Compensable Damages and the Contingent Valuation Method', 34(51) *Natural Resources Journal* 70 (1994); K.E. McConnell,'Reflections on the Ohio Decision', 34(1) *Natural Resources Journal* 93 (1994); K.K. Baker,

its use to exceptional cases where restoration could not be applied (Cross)[95] and those clearly against its employment in litigation because costs outweigh the benefits (Niewijk)[96] or because clearly flawed (Cummings & Harrison, Bohm, Binger *et al*).[97] The former emphasized the advantages of CV (the most complete technique to monetize environmental damage) and the latter its shortcomings (mainly, overestimation of the damage).

In 2002 Thompson made a first review of all cases after the Ohio decision to analyze how much economic evidence was introduced in litigation. Broadly speaking, every time that Courts had to decide on the validity of economic evidence on the non-use value of nature, they were more inclined to accept evidence based on restoration costs rather than contingent valuation.

Very few cases after the Exxon Valdez relied on market-based techniques, including the well-known *California v. BP America (American Trader)*[98] that occurred in the Californian bay on 7 February 1990. There, the lost use value of Californian beaches was awarded by the jury by means of the travel cost approach and by applying the estimations of beaches in Florida.

In other cases,[99] when instead the restoration-cost approach could not be applied because the environment was irreversibly damaged, the Court accepted the Habitat Equivalency Analysis (or HEA) that measures the costs to restore natural resources and services 'equivalent' to the injured ones (see chapter III). These cases show that when NRD claims regard non-use values of nature, a restoration approach was more frequently implemented by judges, whereas non-market valuation techniques run into problems in litigation. When Courts sometimes accept them, they run into issues linked to their validity.[100]

It is very likely that this is the reason why CV has been rarely applied after the Ohio decision and until the early 2000s.[101] For instance, in *Southern Refrigerated,*[102] the State claimed damages for

'Consorting with Forests: Rethinking Our Relationship to Natural Resources and How We Should Value Their Loss', 22(4) *Ecology Law Quarterly* 677 (1995).

95 F.B. Cross, 'Natural Resource Damage Valuation', 42 *Vanderbilt Law Review* 269 (1989).

96 R.K. Niewijk,'Ask a Silly Question ...: Contingent Valuation of Natural Resource Damages', 105(8) *Harvard Law Review* 1981 (1992).

97 R. G. Cummings & G.W. Harrison, 'Was the Ohio Court Well Informed in its Assessment of the Accuracy of the Contingent Valuation Method?', 34(1) *Natural Resources Journal* 1 (1994); P. Bohm,'CVM Spells Responses to Hypothetical Questions', 34(1) *Natural Resources Journal* 37 (1994); B.R. Binger, R. Copple & E. Hoffman, 'The Use of Contingent Valuation Methodology in Natural Resource Damage Assessments: Legal Fact and Economic Fiction', 89(3) *Northwestern University Law Review* 1029 (1995).

98 Case n. 64 63 39 (Cal. Super. Ct. Dec. 8, 1997).

99 United States v. Fisher (Fisher I), 22 F.3d 262, 265 (11th Cir. 1994) and United States v. Fisher (Fisher II), 977 F. Supp., par. 1198.

100 Kopp and Smith examined all the issues of validity that may be raised in litigation when dealing with nonmarket valuation techniques in the famous Eagle Mine case. R. Kopp and V.K. Smith., 'Eagle Mine and Idarado', in K.M. Ward and J.W. Duffield (eds), *Natural Resources Damages: Law and Economics,* at 365 (1992). Particularly, they commented that: 'the level of economic expertise available to judges to evaluate the facts of each side's evidentiary claims probably needs to exceed what many analysts of judicial behaviour have argued can be expected' (at 381).

101 Contingent valuation studies were conducted in several cases but they were all settled, so that judges never ruled on their validity apart from two cases (Thompson, above n. 93, at 78, footnote 42).

102 *Southern Refrigerated,* n. 88-1279, 1991 US Dist. 1869 (D. Idaho 24 January 1991).

water pollution caused by the accidental spill of an agricultural fungicide in the little Salmon River in 1987 and the Court rejected the application of CV because it could not provide estimates with reasonable certainty.[103]

Generally, US judges have been rejecting CV studies because they seemed not to meet certainty standards for scientific evidence. On the other hand, achieving such high standards in litigation is extremely expensive for plaintiffs, so parties might be disincentivized to propose a methodology that will probably be rejected. Interestingly, the same issue of standards came again into the spotlight after the occurrence of the largest oil spill in the US so far: the Deepwater Horizon case (DWH).[104]

In the DWH oil spill, there was a specific request to determine how the 'ecosystem services approach' could help achieve full compensation of damage when valuing post-accident damages[105] and the issue of standards for scientific evidence came back. The case will be analysed in chapter VII after a review of the main ecological literature on the ecosystem services approach. Existing challenges and advantages from this methodology will be illustrated there.

2. EU Law

The second relevant experience on environmental damage assessment can be found in the EU, where the main legislative act providing for an assessment of environmental damage is the European Directive on Environmental Liability (ELD).[106]

Formally, the starting point of the ELD's history can be identified in the year 1986. While the entire Europe was mourning for the accident that recently occurred at the Chernobyl Nuclear Power Plant in Ukraine, another dramatic event happened at the Sandoz agrochemical storehouse in Switzerland causing a tremendous release of toxic pesticides in the air and the underground water. These events raised the level of perceived risk for human health and they ended up in the

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103 Ibid., par. 55-56.

104 See chapter VII, §10.

105 Committee on the Effects of the Deepwater Horizon Mississippi Canyon-252 Oil Spill on Ecosystem Services in the Gulf of Mexico, Ocean Studies Board, Division on Earth and Life Studies, National Research Council, *An Ecosystem Services Approach to Assessing the Impacts of the Deepwater Horizon Oil Spill in the Gulf of Mexico* (Washington DC: The National Academies Press, 2013). As the report pointed out at 1, 'the ecosystem services approach is different from traditional approaches to damage assessment and restoration (e.g., the Natural Resources Damage Assessment, NRDA) because it focuses not on the natural resources themselves, but on the valuable goods and services these resources supply to people. Taking an ecosystem services view can supplement traditional methods of assessing, or valuing, damage to natural resources by estimating flows of goods and services before and after an event. In addition, thinking in terms of ecosystem services would change the way that the public and agencies conceptualize and discuss restoring natural resources to their former condition'.

106 Directive 2004/35/CE of the European Parliament and the Council of 21 April 2004 on environmental liability with regard to the prevention and remedying of environmental damage, OJ L 143/56. The Directive entered into force on 30 April 2007.

resolution of 24 November 1986 of the Council.[107] With this act, the Ministries asked the Commission to investigate the consequences of environmental harm and to review existing measures to prevent and remediate environmental harm. As a response, the Commission adopted its first Proposal for a Directive on civil liability for environmental damage caused by waste in 1989.[108] Among its primary objectives, the 'polluter-pays' principle was mentioned together with the accomplishment of the internal market, the fair compensation of victims and the internalization of waste-related costs.[109] The novelty of the proposal was a liability regime for 'injury to the environment'[110] and not just for traditional damage to persons and property. Soon, the initial project was replaced by a more ambitious one which was not limited to the injury caused by waste.

On 14 May 1993, the Commission published the Green Paper on Remedying Environmental Damage[111] that summarised the main issues before drafting a new piece of legislation. At the same time, in June 1993, the Council of Europe adopted the 'Lugano Convention'.[112] That was followed by a resolution of the EU Parliament asking for a Directive on civil liability for environmental damage,[113] a Working Paper on Environmental Liability in 1997,[114] a White Paper on Environmental Liability in 2000,[115] another Working Paper in 2001 and a proposal for a Directive in 2002.[116]

Finally, on 21 April 2004 the Presidents of the European Parliament and the Council finally signed the text of the ELD and it entered into force in 2007.

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[107] The reference to the Council's Resolution is at p. 1 of the Commission's Proposal of 1991 (*infra* note 52). At that time the term 'Council' unambiguously referred to the Council of Ministers of the EEC (European Economic Community). Following the creation of the European Union with the Maastricht Treaty of 1992, the Council was formally renamed 'Council of the European Union' and it has to be distinguished from the 'European Council' that remains a separate institution devoid of legislative powers and based on intergovernmental decision-making. The Lisbon Treaty officially enlisted it among the EU institutions.

[108] European Commission, *Proposal for a Council Directive on Civil Liability for Damage caused by Waste* [1989] COM (89) 282, amended by [1991] COM (91) 219. The draft discussed the role of civil tort liability for environmental damage.

[109] *Ibid.,* at 1, (2).

[110] *Ibid.,* at 3, (5). It should be noted that the original scope of the Proposal included the three categories of damage to individuals (physical injury, death), damage to property (deterioration, destruction) and injury to the environment.

[111] European Commission, Communication from the Commission to the Council and Parliament and the Economic and Social Committee: Green Paper on Remedying Environmental Damage COM (93) 47 final, 14 May 1993.

[112] Council of Europe, Convention on Civil Liability for Damage resulting from Activities Dangerous to the Environment, 21 June 1993.

[113] European Parliament, Resolution A3-0232/94 of 20 April 1994 on Preventing and Remedying Environmental Damage, OJ C 128, 9 May 1994, at 184-185.

[114] European Commission, *Working Paper on Environmental Liability*, Brussels, 17 November 1997.

[115] European Commission, *White Paper on Environmental Liability* COM (2000) 66 final, 9 February 2000.

[116] European Commission, Proposal for a Directive of the European Parliament and of the Council on environmental liability with regard to the prevention and remedying of environmental damage, COM(2002) 17 final, OJ C 151, 25 June 2002.

The transposition of the Directive has been quite divergent across the EU[117] and the ELD is not applied uniformly among the Member States.[118] Moreover, it has been rarely applied in the EU. A Report by the European Commission in 2016 illustrated that from 2007 (entry into force) to 2013, 11 Member States did not report any ELD case and more than 86% of the whole reported damage occurred in just two Member States.[119] The majority of cases concerned soil pollution (50%) and a minor percentage related to biodiversity damage (20%). According to Lipton and others (2018), various reasons may explain such a limited application of the directive in the Member States.[120] The most important is the perceived complexity, novelty of its content combined with the high costs of enforcement. For instance, notions like the threshold for 'severe damage' are not so clear. Moreover, many competent authorities are not aware of the ELD.[121]

2.1 The key features of the ELD

Regarding the key features of the ELD, four aspects need to be pointed out:

1. the public/private law domain;
2. the liability regime;
3. the scope of the law;
4. the remedies.

First, the ELD did not establish a civil liability regime enabling private parties to sue for damages. It set down an administrative law regime that empowers public authorities to impose specific obligations on polluters in case of imminent threat or already occurred damage to the environment. For this reason, it would be more correct to say that the ELD belongs to the public law area rather than private law.[122]

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[117] BIO Intelligence Service et al. (2013). *Implementation Challenges and Obstacles of the Environmental Liability Directive,* Final Report prepared for the European Commission–DG Environment, p. 21ss.

[118] Stevens and Bolton LLP (2013). *The Study on Analysis of Integrating the ELD into 11 national legal frameworks,* Final Report prepared for the European Commission–DG Environment, pp.34-103; BIO Intelligence Service et al. (2014). ELD Effectiveness: Scope and Exceptions, Final Report prepared for the European Commission–DG Environment.

[119] European Commission (2016) Report from the Commission and the European Parliament under Article 18(2) of Directive 2004/35/EC on environmental liability with regard to the prevention and remedying of environmental damage, Brussels 14.4.2016, COM/2016/0204 final.

[120] J. Lipton, E. Özdemiroğlu Ece, D. Chapman & J. Peers (eds), *Equivalency Methods For Environmental Liability : Assessing Damage And Compensation Under The European Environmental Liability Directive*, at 5 and 18 (2019).

[121] For this reason, the EU Commission has made available training material on its website. See: https://environment.ec.europa.eu/law-and-governance/compliance-assurance/environmental-liability_en#studies

[122] This is a quite common observation that can be found, *ex multis,* in G. Van Calster, L. Reins, 'The Environmental Liability Directive's Background', in L. Bergkamp and B.J. Goldsmith (eds), *The EU Environmental Liability Directive : A Commentary* (2013).

Secondly, on the regime of liability, the ELD opted for a double regime: strict liability for dangerous or potentially dangerous activities (listed in Annex III) and fault or negligence for the others (activities not perceived to be dangerous under Article 3.1). Liability is imposed on the so-called 'operators of occupational activities', where 'operator' refers to the natural or legal person that operates, controls or even exercises decisive economic power over the technical functioning of an activity and 'occupational activity' is defined as any economic activity, a business or an undertaking regardless its private or public, profit or non-profit purpose (Articles 2.6 and 2.7 of the ELD). If the activity is listed in Annex III, then a regime of strict liability applies. Conversely, operators of non-listed activities may be liable upon proof of negligence.

Thirdly, on the scope, for the first time the category of damage to nature or, more in general, to natural resources was legally recognised at the European level. Indeed, it is clearly stated that the Directive does not cover traditional damages granted under international agreements on civil liability or under national civil law regulating personal injury, damage to private property or economic loss.[123] On the other hand, the notion of environmental damage set out in the Directive neglects whether the harmed environment is publicly or privately owned.[124] However, for a number of reasons, the Directive applies to more limited damages, namely to:[125]

- 'environmental damage', meaning 'a significant adverse effect on reaching or maintaining the favourable conservation status of protected species and natural habitats';[126]
- 'water damage', meaning 'that significantly adversely affects the ecological, chemical or quantitative status or the ecological potential of the waters (…) and the marine waters';[127]

[123] Article 4 of the ELD.

[124] Apparently, such a broad definition of ecological damage was already laid down by the Environment Commissioner Ritt Bjerregaard in a Communication regarding environmental liability in 1995. See V. Fogleman, 'The Threshold For Liability For Ecological Damage In The EU', in C. Born, A. Cliquet, H. Schoukens, D. Misonne and G. van Hoorick (eds), *The Habitats Directive in its EU Environmental Law Context: European Nature's Best Hope?,* at 182, footnote 8 (2015).

[125] But Member States can expand its scope beyond habitats and protected species.

[126] Article 2.1.a of the ELD.

[127] Article 2.1.b of the ELD. More precisely, the ELD refers to surface and underground waters covered by the Water Framework Directive or WFD (Directive 2000/60/EC establishing a framework for Community action in the field of water policy, OJ 2000 L327/1). The main aim of the WFD was to reduce water pollution, protect and improve aquatic systems. It applies to all types of water resources. However, regarding marine waters, the directive originally covered only coastal waters. After the huge accident of the Deepwater Horizon in 2010, the scope of the ELD extended to all marine waters under the jurisdiction of Member States, including the Exclusive Economic Zones.

- ‘land damage’, meaning ‘land contamination that creates a significant risk of human health being adversely affected as a result of the direct or indirect introduction in, on or under land, of substances, preparations, organisms and micro-organisms’.[128]

A number of situations remain out of the scope of the ELD,[129] like the damage covered by international civil liability conventions,[130] the damage caused by a third party (provided that the polluter adopted appropriate safety measures) and the damage caused while complying with an order or instruction of the public authority.[131] Member States can add an exemption if the likelihood of the damage could not be foreseen according to the state of scientific and technological knowledge at the time of the polluting event.[132] Lastly, the ELD does not apply to damages caused by events happening before 30 April 2007,[133] not to diffuse pollution (Art. 4.5).

Fourthly, on the remedies, the ELD gives clear priority to restoration (obligation to do) rather than monetary compensation (obligation to pay). From this point of view, the Directive deliberately mirrored the US law on natural resource damage assessment that imposed on liable parties three categories of costs: the costs of restoring the impaired ecosystem to baseline conditions, the interim losses during the restoration period and the costs of assessing damages (administrative costs, costs of enforcement, data collection and monitoring). The differences will be better analysed later in this chapter. However, a prior clarification of the goals of the ELD is needed.

2.2 Efficient deterrence through restoration

‘There is thus quite a significant environmental problem which in great part arose because in most Member States liability for environmental damage has only recently been enacted

[128] Article 2.1.c of the ELD.
[129] Article 4 of the ELD.
[130] E.g., the International Convention of 27 November 1992 on Civil Liability of Oil Pollution Damage. The mentioned Convention covers environmental damage caused by sea-going vessels for oil carriage and it prevails regardless that the ELD imposes more stringent obligations to prevent and remedy the damage. See E.H.P. Brans, *Liability for Damage to Public Natural Resources: Standing, Damage and Damage Assessment*, International Environment Law and Policy Series, Vol. 61 (2001). This even if the damage was caused to a protected area designated under the Habitats and Wild Birds Directives. That happened for instance with the ERIKA oil spill in France in 1999, when the oil affected 400 km of shoreline including Natura 2000 sites.
[131] Article 8 of the ELD. Yet, the European Court of Justice clarified that the ELD prohibits the introduction of national laws with general exemptions from the application of the ELD (general regulatory compliance defence) if the damage is covered by a public authorization (ECJ C-529/15, para. 42).
[132] Article 8(4)(b) of the ELD. These options for exemption resulted in a not uniform transposition of the directive across the EU and, in turn, they led to major problems in dealing with transboundary damage.
[133] Article 17 of the ELD.

(...) Liability should in the future ensure that those who contaminate, clean-up the pollution or pay for the clean-up and, by doing so, encourage (more) socially-efficient prevention by potentially liable parties.'[134]

The roots of the ELD can be found in the EU nature conservation laws and, namely, in the Habitats Directive (HD)[135] and the Wild Birds Directive (WBD).[136] The HD and the WBD were adopted to halt the loss of biodiversity in the EU by forcing Member States to adopt measures to maintain and restore protected species and habitats at a 'favourable conservation status'.[137] Yet, none of these directives included liability provisions to apply the polluter-pays-principle and oblige private parties to reimburse public authorities with the costs of remedial measures, nor had Member States similar provisions at their domestic level.[138]

Given the high rate of biodiversity loss throughout the EU, the ELD was introduced to strengthen the efforts of the previous directives[139] and provide a complementary tool to achieve biodiversity protection through the deterrent effect of liability.[140] In fact, the obligation to prevent and remedy the environmental damage could create an incentive to invest in care *ex ante* to avoid[141] the payment of damages *ex post*.[142] For this reason, when analysing the

[134] Explanatory Memorandum to the 2002 Proposal for a EU directive on environmental liability, COM(2002)17, par.3, summarizing the specific reason for a regime of environmental liability.

[135] Council Directive 1992/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora.

[136] Directive 2009/147/EC of the European Parliament and of the Council of 30 November 2009 on the conservation of wild birds.

[137] Article 2 of the HD.

[138] 'The two main Community legal instruments dedicated to the protection of biodiversity are the Habitats and the Wild Birds Directives. These directives lack liability provisions applying the polluter pays principle and thus encouraging efficient preventive behaviour by private (and public) parties. Currently few, if any, Member States fill this void by imposing liability for biodiversity damage on private parties. Thus, Community action to protect and restore biodiversity is warranted on two main grounds: ensuring socially-efficient means are used to finance the remedying of damage to biodiversity in the Community and, by doing so, encourage efficient prevention." (Explanatory Memorandum to COM(2002)17, par.3).

[139] The definition of ecological damage in the current consolidated version of the ELD is the result of a process of convergence between the first communications on environmental liability and the Habitats and Wild Birds Directives. While the first documents adopted a broad notion, the latter progressively limited the scope of EU environmental liability. The Draft White Paper on Environmental Liability of 14 September 1998, 10, s. 6.5.1, shows their influence: 'damage to biodiversity as protected under the Habitats and Wild Birds Directives should be covered'. The White Paper on Environmental Liability of 2000 (COM(2000) 66 final, 9 February 2000, 12, s 3.3) was even more explicit: 'important that an EC environmental liability regime should also cover damage afflicted upon natural resources, at least those that are already protected by EC law, namely under the Wild Birds and Habitats Directive, in the designated areas of the Natura 2000 network'. However, the Commission proposed to limit the scope of the ELD to the so-called 'biodiversity damage' just for an initial phase and to ensure optimal legal certainty, considering that a regime of environmental liability was new in many of the Member States (Communication from the Commission to the Council and the European Parliament – Biodiversity Action Plan for the Conservation of Natural Resources COM(2001) 162 final, 27 March 2001*,* par. 76). Later, in 2001, the scope was again extended to include damage to sites protected by national nature conservation laws.

[140] The specific mechanism of deterrence can be summarised in the following way: 'The prevention and remedying of environmental damage should be implemented through the furtherance of the principle according to which the polluter should pay, as indicated in Article 174(2) of the Treaty. One of the fundamental principles of this Directive should therefore be that an operator whose activity has caused the environmental damage or the imminent threat of such damage will be held financially liable in order to induce operators to adopt measures and develop practices to minimise the risks of environmental damage so that their exposure to financial liabilities is reduced.' (2002 Proposal for a Directive, above n. 112, par. (2)).

[141] 'This will give people carrying out activities that risk damaging protected natural resources additional incentives to take appropriate measures to avoid problems.' (Biodiversity Action Plan 2001, above n. 136, par. 78).

[142] 'The liability mechanism moreover has the merit of internalizing environmental costs, as eventual restoration costs may be considered by, for example, insurance companies when setting premiums'.(ibid.)

remedies under the Directive, it is important to bear in mind that the Directive is not only aimed at restoring the injured environment (covering the cost of restoration) but also at providing polluters with optimal care incentives if they know that they have to pay for the full cost of accidents.

2.3 The procedure under the ELD

Under the ELD, the operator has to notify the competent authority that environmental damage has occurred (or that an imminent threat exists) and to take measures to prevent and remedy the environmental damage.[143] After the notification, the public authority establishes which measures have to be implemented and it checks the threshold (significance of damage).[144]

The process to choose the appropriate measures unfolds in this way:[145]

a) first step is to set restoration targets;
b) second step is to identify restoration options (choosing between no intervention, limited intervention, and full-scale reconstruction);
c) third step is to scale restoration options through an evaluation process that weighs the cost of each option, the time for restoration to be effective, the extent to which each option will prevent future damage, other benefits for the environment and the public health.

The aim is to select the least costly option that leads to the restoration targets through a process known as cost-effectiveness analysis (CEA).[146] The cost of each option includes the costs of damage assessment and those to implement restoration (cleaning and restoring species, habitats). They need to be weighed against the benefits of restoration (in terms of ability of damaged resources to provide services) to establish whether a restoration option is cost-effective and that it can be implemented.

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[143] Under Article 6 of the ELD: "the operator shall, without delay, inform the competent authority of all relevant aspects of the situation and take: (a) all practicable steps to immediately control, contain, remove or otherwise manage the relevant contaminants and/or any other damage factors in order to limit or to prevent further environmental damage and adverse effects on human health or further impairment of services and (b) the necessary remedial measures, in accordance with Article 7".

[144] The main condition for applying the ELD is the occurrence (or the threat) of 'significant damage' to the covered natural resources. The threshold for the damage to protected species and natural habitats is achieved if the damage has 'significant adverse effects on reaching and maintaining the favourable conservation status'.(Article 2.1(a) of the ELD). For the definition of 'conservation status' of habitats and species, see Articles 2.4(a) and 2.4(b) of the ELD. For instance, the conservation status of the habitat is: 'the sum of the influences acting on a natural habitat and its typical species that may affect its long-term natural distribution, structure and functions as well as the long-term survival of its typical species' (Article 2.4(a) of the ELD).

[145] EU Commission, Directorate-General Environment, "Study on the valuation and restoration of damage to natural resources for the purpose of environmental liability", B4-3040/2000/265781/MAR/B3, Final report by Macalister Elliott and Partners Ltd and the Economics for the Environment Consultancy Ltd, 2001.

[146] 'The ideal outcome of a liability regime would be a solution that provides full compensation to the public for damages to natural resources at the least cost to the liable party' (ibid, at 3).

The competent authority might also intervene and take the necessary remedial measures, 'as a case of last resort, when the operator fails to comply with the obligations or cannot be identified or is not required to bear the costs under the Directive'.[147]

In this case, 'the competent authority shall recover, inter alia, via security over property or other appropriate guarantees from the operator who has caused the damage or the imminent threat of damage, the costs it has incurred in relation to the preventive or remedial actions taken under this Directive.', unless 'the expenditure required to do so would be greater than the recoverable sum or where the operator cannot be identified'.[148]

There is no ceiling for liability[149] but that does not mean that liability under the ELD is unlimited. Restoration measures need to be in any case reasonable under the directive (restoration cannot be disproportionately costly).[150] Yet, the threshold where the costs of restoration become disproportionate is not defined.

2.4 The remedies

After the occurrence of the environmental damage, the liable party has basically to pay for the restoration of the environment. The ELD implements the polluter pays principle by forcing the polluter to pay for the costs of remedial and preventive measures.

The method of environmental damage assessment under the ELD has been largely debated. In the end, restoration as primary remedy was chosen because: 'restoration costs are easier to estimate, rely on fewer untested economic valuation methodologies, and are verifiable ex post'.[151] Moreover, it allows to adopt equivalent alternatives to the damaged natural resources rather than replicating them. Lastly, it overcomes the risk that monetary compensation is not spent on returning the environment back to the baseline.[152]

More precisely, three remedial measures are set down by the ELD:

[147] Article 6.3 of the ELD.

[148] Article 8.2 of the ELD.

[149] The lack of ceiling raises an issue for introducing insurance. For this reason, the EU Commission explored other tools like funds and risk-pooling schemes (see Lipton et al., above n. 120, at 8, footnote 21).

[150] Annex II, par. 1.3.1, ELD.

[151] Explanatory Memorandum to COM(2002)17, par.6.

[152] in 1995, Commissioner Bjerregaard explicitly referred to the risk that damages were not spent on restoration: 'It has to be ensured that, when someone has to pay damages for the restoration of ecological damage, the money is actually spent to restore the damage or, if this is factually impossible, to bring equivalent elements into the environment'. See the Communication to the Commission on Community Action as regards Environmental Liability (December 1995) 11, s VI(2). This early statement reveals the preference of the European Commission for restoration already when discussing the first drafts of the ELD.

- primary remediation: 'any remedial measure which returns the damaged natural resources and/or impaired services to, or towards, baseline condition';[153]
- complementary remediation: 'any remedial measure taken in relation to natural resources and/or services to compensate for the fact that primary remediation does not result in fully restoring the damaged natural resources and/or services';[154]
- compensatory remediation: 'any action taken to compensate for interim losses of natural resources and/or services that occur from the date of damage occurring until primary remediation has achieved its full effect'.[155]

Compensation of interim losses is needed because it takes time to restore damaged natural resources to the baseline and the polluter is also liable for the lost resources and services until full restoration. Moreover, the polluter is liable for preventive and remedial costs incurred.[156]
Another issue to point out is that the aim of all remediation measures is to restore the damaged resources and also their services, defined as the 'functions performed by a natural resource for the benefit of another natural resource or the public'.[157] For instance, a damaged coast provides food for birds, clean water for fish, recreational activities to the people (beach use, boating, etc.), wildlife viewing, hunting. The mention of services reflects the increasing relevance of the ecosystem services by the time the directive was adopted[158] and it marks a crucial distinction between the ELD and the other nature conservation directives (Habitats and Wild Birds Directives) that do not refer to human benefits from protected species and habitats even when talking of compensatory measures.[159]

2.5 Obstacles for full restoration

There are a number of issues that might prevent from achieving full restoration as primary remedy:

- costs of information and enforcement for public authorities;
- lack of guidelines on primary restoration;

[153] Annex II of the ELD, par. 1(a).
[154] Annex II of the ELD, par. 1(b).
[155] Annex II of the ELD, par. 1(c).
[156] Article 8.1 of the ELD states: "The operator shall bear the costs for the preventive and remedial actions taken pursuant to this Directive".
[157] Article 2.13 of the ELD.
[158] The Millennium Ecosystem Assessment was initiated in 2001 on the request of the United Nations Secretary-General Kofi Annan in order to assess the effects of the ecosystem changes on human beings and it led in 2005 to the release of five technical volumes and six synthesis reports on the state of ecosystem services in the world. The point of the evolution of the language of liability laws has been raised also by Jones and DiPinto in US law (see *infra*).
[159] See Article 6(4) of the HD.

- lack of precise time constraints;
- irremediable damage;
- impossibility to identify the liable party.

2.5.1 Costs of information and enforcement for public authorities

Based on the most recent report on the implementation of the ELD, it seems that very often authorities begin procedures under the Directive *ex officio.* Cases starting upon notification of the polluter (through self-reporting)[160] are pretty rare.[161] When the authorities are the primary parties who activate the procedure, it is because they get the related information during their monitoring activities, such as inspections on waste or industrial sites, surveys on polluting activities, etc. Apparently, relevant authorities usually discover environmental accidents while enforcing national sectorial laws rather than national ELD laws (and often they are not the authorities competent for the ELD but rather forestry, water protection bodies, municipalities, policemen and prosecutors working on environmental crimes). Another reason for the scarcity of systematic environmental inspections is a very basic one and it refers to the lack of staff within public authorities designated as competent on the ELD. Indeed, the employees of competent authorities often lack capacity and ability to search cases of environmental damage under the directive.[162] This leads to the need for more capacity building activities. This means that the Directive does not provide competent public authorities with sufficient incentives to enforce it by conducting related inspections and the discovery of biodiversity cases relies on the cooperation between them and the other national authorities competent on sectoral laws (this would be probably the cheapest way since it would employ already well-trained personnel).[163] Yet, even when well-informed, national researchers from various EU countries reported that 'the authorities may be afraid to initiate complicated ELD processes and risk that the process will not

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[160] An interesting effect which is often overlooked is the choice of law upon which the polluter's notification can be based. In many countries, polluters can report an existing environmental damage (or threat) pursuant either national laws implementing the ELD or national sectorial laws. Often, the ambiguity of definitions within the former combined with the higher magnitude of expected liability (e.g., remedial action), induce polluters to self-report under sectorial laws which embed clearer definitions of environmental damage and that only oblige polluters to pay a fine for pollution. Clearly, lack of coordination between competent authorities in EU countries and vague norms in the ELD undermine the achievement of optimal care incentives on potential polluters. To overcome this issue, it was proposed to introduce an obligation of sectoral authorities of reporting the notice of the accident to the competent authority under the ELD. See: A. Thomas, G. Schamschula, B. Schmidhuber, F. Bouquelle, L. Lavrysen, *et al.,* 'Improving Implementation and the Evidence Base for the ELD', at 131 (2021). Additional ideas would be to reduce transaction costs of polluters when they have to self-report, e.g. apps to inform the public authorities in a quick and cheap manner.

[161] 2021 Report, above n. 160, at 126-127. According to the 2021 Report, the only country where most of the ELD cases were communicated by the polluters is Portugal.

[162] Evidence on this point has been collected especially in Greece (2021 Report, above n. 160, at 132).

[163] Ibid., at 127.

lead to the determination of liability under the ELD, which would thwart all their work'.[164] In other words, authorities willing to start ELD procedures have to weigh the probability of identifying the polluter and getting reimbursed against the costs of the whole procedure. If the latter are higher than the former, authorities (that are not obliged to start the procedure) might forego a ELD procedure. Few countries provide mandatory intervention by competent authorities.[165] For instance, in Latvia, Luxembourg, Slovakia, Romania and Spain, competent authorities have to intervene when the polluter is not carrying out any or sufficient preventive measures, when he fails to comply with the instructions given by the environmental bodies and when he cannot be identified.[166] In Portugal, the environmental authority may intervene under all these circumstances, but intervention becomes obligatory if the magnitude of the damage is very high or there is harm to property and people (but the expenses shall be recovered from the polluter).[167]

2.5.2 Lack of guidelines on primary restoration

In case of actual environmental damage or imminent threat of damage under the directive, the primary obligation to take remedial measures lies upon the polluter for the simple reason that he is in the position to best know when the damage occurs and which measures are most effective. Therefore, the polluter is expected to take urgent measures aimed at preventing further damage (expansion of the damage) or avoid raising the risk of environmental damage (the threat of damage needs to be removed in the fastest way). Likewise, the polluter should refrain from any activities that would create further threat. After that, the polluter should plan the remedies to tackle the consequences of pollution (a clean-up programme). Many countries, such as Italy, Latvia and Czech Republic, impose on the polluter the obligation to set down a clean-up programme and send it for approval to the competent authorities within a certain time.

Some of the issues that might undermine the results of the remedial actions undertaken by polluters are the absence of detailed guidelines for the remedial action plan, (e.g., on the costs and the likelihood of success of restoration),[168] the low level of scrutiny over recovery programmes (which is intertwisted with the expertise of those in charge of monitoring activities) and the lack of proper knowledge and equipment by the liable party. For instance, if a fire occurs

[164] Ibid., at 130.
[165] According to the 2021 Report, above n. 160, at 138, cases of mandatory public remedial action are limited because "environmental authorities are usually too short of resources".
[166] Ibid., at 137.
[167] Ibid., at 138.
[168] 2021 Report, above n. 160, at 136ss. Apparently, Estonian authorities do provide polluters with guidelines.

in an industry, it is possible that the owner of the industry is not well equipped to tackle it and urgent remedial action by the public authority would be more appropriate (cheaper and more timely). Another very crucial issue is that liable parties obliged to undertake restoration have a private interest in investing as little as possible in restoration.[169] Public officials willing to counteract this strategic behaviour would need evidence about the objective magnitude of environmental damage. Yet, data on the state of the environment prior to the damage might be unavailable or too costly to be collected.

Regarding the types of remedial measures that can be ordered by the competent authority to the operator, below there is a list of possible types found throughout the EU:[170]

- measures to limit or cease the polluting activity are the most practical and effective (very few reported in Cyprus).
- measures aimed at an objectively fixed result, such as restoring a river to the baseline, excavating a soil for decontamination, removing substances, removing organisms, reducing the concentration of substances to a level that is not seriously risky for people.
- measures to provide the authority with information (measurements, monitoring) in order to follow up the remediation.
- measures to control and stop the transfer of polluted property areas.

Lastly, given that remedies have to be decided based on certain criteria (their costs, probability of success, prevention of future environmental damage, time of reparation, integration of other factors),[171] a certain degree of discretion in the decision-making is unavoidable. Moreover, in soil clean-up, the objectives of remediation are complemented by regional laws which might identify different values for soil remediation.[172]

Evidence has also been collected showing that in many countries poor or no remedies have been adopted even when the competent authority was notified of the imminent threat to the environment (neither primary, compensatory, or complementary remediation).

2.5.3 Lack of precise time constraints

Generally, there are no statistics and data available on the time dimensions of ELD cases in the EU and the timeline depends on several factors: the availability of good information about the

[169] Ibid., at 137 with a reference to the opinion collected from the officials of the Lithuanian Environmental Protection Department. Liable persons are mostly ‘unwilling to fully restore the environment; they try to restore it as little as possible. They often challenge instructions, requests, or orders from officials in this regard’ (at 137).
[170] Ibid., at 147. The listed measures have been reported especially in Czech Republic and Sweden.
[171] ELD, XX.
[172] 2021 Report, at 148.

damage, the cost of decontamination, the existence of an addressee, the capacity and competence of the authority, the workload in general, the way the case is prioritised in relation to other cases, the need for urgent measures, the lack of precedents or clear guidelines on difficult cases.[173] Although in some countries polluters have to notify the accident to competent authorities within short timeframes (Belgium, Italy, Portugal, Romania) and the remediation itself must be concluded within a given deadline (Latvia, Belgium), the practical implementation of remedies often takes more time and legal procedures last even longer.[174] In Estonia, it has been reported that 'the time period from registration of the case to identification of the measure may range from one week to one year, depending on the circumstances. Application of the measures takes even more time, not to mention restoration of the environment to the original condition'[175]. In Portugal, for instance, the last Court hearing of a complex case (*Siderurgia National*) which started in the 1970s took place only fifty years after the accident.[176]

The delay might be due to various factors. One of these might be that more than one authority can be competent for preventive measures (e.g., in Germany several federal authorities might have competence) and the decision-making tends to get longer. Other possible reasons for delays are the relative rarity of ELD cases, the shortage of officials in charge of carrying out investigations in the administrative authorities, and the lack of accelerated procedures and simple rules. A Danish researcher also pointed out a paradox of the ELD that is plausibly the main obstacle to timing emergency responses: under the ELD, the polluter is obliged either to take the needed preventive measures to avoid the damage or to take any measure to limit the extent of the damage straight after the accident.[177] The contradiction arises if the responsible party cannot be identified quickly or it is unsure whether the damage falls under the ELD. Realistically, especially in case of largely polluting events, it takes a lot of time to clarify the facts, identify liable parties and to take clean-up decisions carefully through administrative procedures. In all these cases, if the wrong person is selected as liable party or the liable party is obliged to carry out unnecessary remediation, the public authority might be required later to pay back large amounts of money for wrongly assigned legal and financial responsibilities.[178] Therefore, competent authorities implementing emergency remediation by the time administrative procedures take place, are

[173] 2021 Report, at 174.
[174] 2021 Report, at 176. The mentioned cases last between five and nine years after the accident was detected. Delays have been reported more or less everywhere in Europe: Finland, Belgium, Slovakia, Denmark, Germany, Sweden, Estonia, Portugal. As warned by the Report, it does not make sense to establish the average length of cases only based on the largest ones.
[175] 2021 Report, above n. 157, at 177.
[176] 2021 Report, above n. 157, at 178.
[177] Ibid.
[178] 2021 Report, above n. 157, at 178-179.

induced to invest as little money as possible on remediation. Indeed, 'if the authority goes too far in the process and spends too much money before having identified the responsible operator, this can lead to the public having to pay for the clean-up. If the perpetrator is not found or cannot pay, the public is left with the whole cost of the clean-up'.[179]

During the time needed to identify the liable person or to determine whether ELD procedures are applicable, the internal situation of polluters (especially companies) is very likely to change (sale of assets and start of bankruptcy procedures) and, so, the identification of polluters becomes more difficult. This is an additional reason for which the costs of remediation spent by public authorities for emergency clean-up might remain born by taxpayers in the end.[180] A solution adopted in Sweden to tackle the change of ownership is to restrict the transfer of polluted lands by prescribing specific precautions. First of all, polluted areas are declared 'environmental hazard zones'; this designation is then noted in the land register and the authority imposes restrictions on the use of the area (obligatory environmental investigations and notifications to the public authority prior to any transfer).[181]

To sum up, time uncertainties and possible delays induce public authorities to carry out quick emergency remediation just to contain pollution, while polluters exploit them to leave the market and escape the final payment. In order to ensure that the environment is brought back to baseline conditions and polluters are correctly incentivized to prevent accidents, it is thus essential that emergency clean-up under uncertainty is accompanied by alternative means of compensation and insurance. For example, in a Belgian road accident that caused a pesticide spill in 2014 in the Walloon Region, the ELD procedure started in 2015 and the environmental authority took a final decision on remediation measures (primary, complementary and compensatory restoration) only in 2019. Given that the polluter was covered by insurance, measures in the preliminary phase were easily executed by the insurance company. Likewise, in the case of mines, measures to clean-up and prevent further damage can take decades, even if decisions are taken in short timeframes.[182] This is mainly due to the fact that mines are quite big and restoration is extremely expensive.

[179] Ibid.
[180] 2021 Report, above n. 157, at 180.
[181] 2021 Report, above n. 157, at 141.
[182] 2021 Report, at 179.

2.5.4 Irremediable damage

There are cases of substances, such as mercury, arsenic, cyanide, chromium, and lead, that pose severe risk for human beings and that are deposited in such large amounts that their removal is simply impossible. The only remediation that is feasible here is to isolate and store them, for instance in old salt quarries, although the geological movement of rocks can create concerns for the safety of underground water.[183] According to the ELD, if it is not possible to restore the environment back to the baseline, the polluter has to complement primary remediation measures with the replacement of the damaged natural resources by equivalent alternatives, of which the cost is equivalent to the monetary value of the lost resources. This needs to be done in other places than the damaged site (see §3ss).

2.5.5 Impossibility to identify the liable party

When pollution happens in sites where there is no liable party because the site has been physically and/or legally abandoned (orphan sites), the polluter pays principle cannot be applied. The issue of orphan sites belongs therefore to the broader topic of historical sites, i.e. sites where pollution has a longstanding nature.[184] These types of old polluted sites pose the issue of whether the States should be in charge of their rehabilitation, as they normally fall out of the scope of national ELD law. Given the availability of limited resources for their remediation, competent authorities usually refrain From conducting full restoration and they only halt further spread of damage or prevent harm to human health.[185] Moreover, detailed investigations of polluted historical and orphan sites are poorly available.[186] The issue of the source of money spent by environmental authorities on the clean-up of historical polluted sites is a central one for the internalization of environmental costs. An interesting example relates to the compensation of environmental harm in the abandoned mine tailings ponds in Spain.[187] In 2019, 44 abandoned sites were counted and the Spanish ELD law set down two funds for compensation: a "Compensation Fund for Environmental Damages" managed by insurers (Consortium of Insurance Compensation) and financed by the insured operators (through their insurance premium), and a "State Fund for the

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[183] Ibid., at 148.
[184] These definitions are provided by the 2021 Report, above n. 157, at 143.
[185] Ibid., at 139.
[186] For instance, in Lithuania only 4.5% of the potential pollution hotspots have been investigated in detail by 2019. By 2020, only 13.2% of the investigated areas had been remediated and mainly with EU funds. Moreover, there are no data on the application of environmental liability laws to these cases. In Greece the only data available for orphan sites are from 2011 to 2015 as they were remediated through a Green Fund under the Urban Rehabilitation Programme. No data are available in Hungary since 2003 and no data in Slovenia (ibid., at 140). Seemingly, data on old pollutes sites exist only for sites that were successfully cleaned-up. The only countries with national programmes to detect and remediate old polluted sites that have been reported are Sweden, Romania, Latvia, Germany.
[187] Ibid., at 141.

Remediation of Environmental Damages”, that covers the remediation of damages caused to natural resources owned by the State (e.g., coasts). The first tool is a mandatory financial guarantee limited to damages occurred during the time of the insurance,[188] while the second finances the costs of preventive, avoidance and restorative measures when operators are not forced to pay for them. According to Article 24 of the Law n. 26/2007, the financial guarantee includes the cost of prevention, avoidance and primary remedy, with no ceiling to the environmental liability of the operator. Moreover, the insurance does cover only environmental liability, separately from other civil, administrative or criminal liabilities.[189]

3. Compensatory restoration in the US and the EU

The previous section listed a number of issues that might prevent from achieving full restoration. Their occurrence (or non-occurrence) may bring to two possible scenarios:

1) None of the issues in § 2.5 occurs and full primary restoration is achieved.
2) At least one of the issues in § 2.5 occurs and full primary restoration is not achieved.

Under scenario (1), ecological studies show that even if injured natural resources are fully restored to the baseline, the society still needs compensation for the losses of natural resources and their services between the moment of the accident and the moment they return to baseline. Under scenario (2), if injured natural resources cannot be fully restored, the society needs compensation for permanent losses.

The activity to address and compensate for interim and permanent losses goes under the name of ‘compensatory restoration’ and the last stage of the environmental damage assessment both in the US and EU law concerns remedial measures serving to compensate the public for the loss of natural resources and services during the recovery period, the so-called interim losses (see §1.4 and §2.4).

[188] It was introduced by the Law n. 26/2007 of 23 October on environmental liability and the Regulations partly implementing Law n. 26/2007 of 23 October on environmental liability (Royal Decree 2090/2008, of 22 December).
[189] But following Article 30 of the Law n. 26/2007, the mandatory financial guarantee cannot exceed € 20.000.000 euros, although this does not exempt operators from covering the full costs of prevention, avoidance and primary remedy.

The process 'to ensure that compensatory restoration neither over-compensates nor under-compensates for service losses'[190] is called 'scaling'.

The aim of this section is to illustrate the main differences between the law and the practice of compensatory restoration in the US and the EU, given its importance for optimal deterrence and cost-effective restoration. First the differences in the law will be illustrated. Then, the different practice in the application of the law will be examined.

3.1 Compensatory restoration in US law

Under US law, there are three main approaches to scale compensatory restoration options.[191]

The first is the 'service-to-service' approach (resource-to-resource method) that is based on a one-to-one trade-off, meaning that the lost service is replaced by a new one that provides the same quantity of services. Whenever possible, public authorities and trustees should apply this approach. If the injured and the replaceable natural resources are of the same type, quality and value, this approach is the most appropriate.

The most common service-to-service approach is the Habitat Equivalency Analysis (HEA). Under US law, the HEA entails three main steps:

1. quantifying the present (discounted) value of service losses;
2. quantifying the present value of service gains (provided by compensatory restoration projects);
3. calculating the quantity of restoration needed to equate losses and gains.[192]

In order to quantify service losses, it is important to take into account: the starting moment of the injury, the percent service level prior to the accident (baseline), the service decline function, the percent service level decrease, the extent of injury (hectares of habitat or number of individual organisms), the starting moment of recovery, the service recovery function (time-path) and the max percent service provision after restoration.

In order to quantify the benefits of compensatory projects, the following factors need to be assessed: the initial percent service of a compensatory site; the starting moment of provision of services in the compensatory site, the service provision function (time-path), the duration of the compensatory project, relative value of the compensatory resource compared to the injured one.

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[190] W.H. Desvousges, N. Gard, H.J. Michael & A.D. Chance, 'Habitat and Resource Equivalency Analysis: A Critical Assessment', 143 *Ecological Economics* 74, at 75 (2018).

[191] The first guidance document on scaling restoration projects under the Oil Pollution Act of 1990 was issued by NOAA in 1997. It recommended to scale restoration through the Habitat Equivalency Analysis and the cost of the project is considered to be the measurement of the damage.

[192] National Oceanic and Atmospheric Administration, Damage Assessment and Restoration Program, 'Habitat Equivalency Analysis: An Overview' at 24 (1995; 2006).

After discounting service gains and losses, restoration is scaled by dividing the number of restoration units needed to compensate the public for lost natural resources by the number of units lost.

If the service-to-service approach is not feasible, i.e. a one-to-one match is not possible, the second scaling approach is the 'value-to-value'. It differs from the previous one because it is based on the economic value (obtained through nonmarket valuation methods) of the damage[193] rather than a physical measurement of the services provided. Its aim is to identify a restoration option such that the benefit of the compensatory option is equal to the economic value of the lost services.[194] In the US, when scaling a restoration action, trustees have to discount all service quantities and/or values to the date of the claim and to evaluate the uncertainties of restoration actions. The criteria to follow when selecting the appropriate restoration action include the capability of returning the resource to baseline in an 'expeditious and cost-effective'[195] manner while involving the interested parties in the administrative process (cooperative approach).

A third possible approach to scaling is the value-to-cost. According to the law, this should be the least preferred, but it is in fact the most common. Here, the value of the lost services and/or resources is weighted against the cost of restoration, instead of measuring the benefits of restoration. While this approach ensures equivalency between the value of the loss and the cost of the compensatory restoration project, there is no warranty that this value is equivalent to the social benefits.[196] Apparently, this is the most common approach in the US[197] because it is convenient for all the parties cooperating in the NRDA: trustees can save funds and staff, the polluter has less assessment costs to reimburse and the public can benefit sooner from the completion of restoration. In other words, the cooperative nature of the NRDA in the US allows to overcome the limits of the more accurate value-to-value approach.

[193] The 'value of damage' to the environment can be achieved through economic valuation techniques that measure public preferences for an environmental state. These techniques include stated preference and revealed preference mechanisms aimed at eliciting people's preferences through surveys, in the first case, or by using data from actual markets, in the second case. By contrast, costs of clean-up and restoration do not need to previously identify a damage and damaged parties. They are based on technical options available rather than on public preferences.

[194] Under US law, trustees have to measure the value of injured natural resources or services and then 'select the restoration action that has a cost equivalent to the lost value' (ibid.).

[195] 15 CFR § 990.10 – Purpose.

[196] On this point, see R.E. Unsworth and R.C. Bishop, 'Assessing Natural Resource Damages Using Environmental Annuities', 11(1) *Ecological Economics* 5 (1994).

[197] S.M. Thur, 'Resolving Oil Pollution Liability With Restoration-based Claims', 32(3/4) *Océanis* 375, at 382 (2006).

3.2 Compensatory restoration in EU law

In the EU, the declared goal of compensatory restoration measures is to address and compensate for the interim losses of natural resources and services until full recovery. [198] Moreover, the primary technique to scale compensatory remedial measures is the 'service-to-service' or 'resource-to-resource' equivalence approach, like in the US.[199] Therefore, actions to provide resources and/or services of the same type, quality and quantity should be given priority.

If these kinds of resources and/or services are not available, alternative natural resources and/or services have to be provided and, for instance, a reduction in quality may be offset by an increase in quantity.[200] The equivalence between the economic value of the loss and the benefit of the alternative restoration option is not mentioned very clearly like under US law.

Lastly, if it is not possible to use the service-to-service approach, the competent authority may prescribe the employment of monetary valuation to scale compensatory measures. If, in this case, the valuation of lost resources/services is practicable, but valuation of replacement resources/services requires excessive time or money ('cannot be performed at a reasonable time-frame or at a reasonable cost'), the competent authority can choose remedial measures of which the cost is equivalent to the monetary value of lost resources or services. [201] This reminds about the third scaling approach in the US (value-to-cost). Moreover, it makes the reader infer that the goal of the monetary valuation should be first to equate the value of the damage with the value of the replacement resources and/or services (value-to-value). Only if this is not practicable, then the value-to-cost comes in.

In light of the above, if one had to respect the sequence of valuations for scaling in the order that appears in the law, the economic valuation of damage would be still needed either in the US or in the EU for the compensation of interim losses. Under EU law, economic valuations are explicitly mentioned as 'alternative valuation techniques' for compensatory restoration. However, neither the wording of the ELD is detailed and precise like in US laws nor the economic valuation is made mandatory.

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[198] It must be kept in mind that interim losses occur over an infinite period of time if primary restoration is not possible. The magnitude of interim losses depends indeed on the primary restoration options and the time for recovery to take place.
[199] ELD, Annex II, par. 1.1.3.
[200] Ibid., par. 1.2.2.
[201] Ibid, par. 1.2.3 and see EU Commission, above, at 2.

3.3 The practice of compensatory restoration in the US

Compensatory restoration started being used in some environmental damage assessments in the US in the 1990s.[202] By the beginning of the 2000s, the 'paradigm shift' was considered completed[203] and in the first years of the 2000 the employment of the HEA became largely widespread in the US practice (when the ELD in the EU was about to be born). On the one hand, US trustees seemed to be much more inclined to use equivalency analyses in order to save time and money[204] and, on the other hand, equivalency analyses facilitated the early conclusion of liability lawsuits by providing a basis for settlements.[205] As notably stated: 'the principle of equivalency analysis may have been lost or ignored in the rush to find a simple method of analysis'.[206] In a few years, the HEA/REA became the primary method for the calculation of environmental damage in the US by the early 2000s.

In 2018, Jones and DiPinto conducted an empirical analysis of US cases on liability for damage to public natural resources and services with non-use values over the 25 years since OPA was promulgated to determine which approach is preferred by US trustees to scale compensatory restoration projects.[207] They found that trustees mostly rely on the service-to-service approach (HEA) rather than what they called, a 'valuation' approach (i.e., surveys). This is mainly because the HEA 'simplifies complex ecosystems through the choice of representative ecological process or function metrics as proxies for the change in the quantity and quality of service levels at the injury and restoration sites in a particular case'.[208] Also, in the few cases that were litigated, Courts usually uphold the HEA.[209]

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[202] See, for instance, Texas General Land Office, Texas Parks And Wildlife Department, Texas Natural Resource Conservation Commission, NOAA, US Fish & Wildlife Service, US Department Of The Interior, 'Draft Damage Assessment and Restoration Plan and Environmental Assessment for the Point Comfort/Lavaca Bay Npl Site Recreational Fishing Service Losses' (1999).

[203] N.E. Flores & J. Thacher, 'Money, Who Needs It? Natural Resource Damage Assessment', 20(2) *Contemporary Economic Policy* 171, at 171 (2002).

[204] Thompson, above n. 93.

[205] On the website of the US National Oceanic and Atmospheric Administration (NOAA) , it is possible to find many cases of oil spills, hazardous waste sites and ship groundings where the compensation of natural resource damage was settled between the public authority, the trustees and the polluters. See: https://darrp.noaa.gov/what-we-do-resources/explore-cases [accessed 20 November 2023].

[206] See W. H. Desvousges, N. W. Gard, H. Michael, A. D. Chance, 'Habitat and Resource Equivalency Analysis: A Critical Assessment', in 143 *Ecological Economics* 74 (2018), at 75.

[207] C.A. Jones and L. DiPinto, 'The Role of Ecosystem Services in USA Natural Resource Liability Litigation', 29 *Ecological Services* 333 (2018). The study is quite relevant given that Carol Adaire Jones was lead economist on the Oil Pollution Act regulations from 1990 to 1997 overseeing 36 claims for NRDA including the Exxon Valdez case and Lisa DiPinto is current senior scientist for NOAA's Office of Response and Restoration., plus coordinator for the Deepwater Horizon case of 2010.

[208] Jones and DiPinto, above n. 207, at 340.

[209] For instance, *US v. Fisher* et al. 1997 and *US v. Great Lakes Dredge and Dock Co.* et al. 1999. In both cases, federal courts awarded damages for the destruction of acres of damaged seagrass. More recently, in 2008 and 2012, the use of HEA was upheld for valuing forest fire damages in US public land.

Precisely, NOAA started by using the HEA for the calculation of environmental damage in the early 1990s for ship-grounded cases in Florida where small quantities of coral reefs and sea grass were injured. Subsequently, trustees extended the use of the HEA to difficult-to-value environmental damage caused by oil spills, hazardous material releases, forest fires and to a wide range of habitats, including wetlands, rivers, beaches, fish, aquatic birds and endangered species. The simplicity and flexibility of the HEA allowed to speed up the assessment and to quickly provide a basis for restoration settlements, which is how most of the cases end up in the US. Parameters for measuring the service losses tend to be much more simplified: the baseline is assumed as 100%, the decline function is often taken as instantaneous, the extent of the injury is measured, while the restoration function and the service provision are the object of subjective decisions based on previous restoration projects.[210]

In one of the larger settlements for hazardous waste pollution so far in the US, the Blackbird mine case study (contamination from the Blackbird copper mine), trustees decided to conduct a cost-effective damage assessment to quantify the injury and establish the restoration goals. They thus focused on just three ecological metrics: quality of surface waters, injury to food web and injury to Chinook salmons (endangered species). Based on them, two restoration goals were identified: restoration of salmon population and restoration of fishery habitat for compensatory restoration. The recovery of the salmon population at the ‘carrying capacity’, i.e. 100% service level, would be an indicator of full recovery also of other indicators, such as the water quality. Plus, salmons would guarantee the return of nutrients for the whole stream, recreational and cultural services related to the fishery. The HEA was instead used to calculate the amount of required fishery habitat to compensate for interim losses in the same location of the injury and with comparable size.

To favour appropriate scaling of compensatory restoration, NOAA supported the production of reports, concepts, models and techniques developed over the past decades since the early 1990s.[211] These documents provide reference to case studies and primary ecological scholarship and their primary aim is to provide guidance for restoration scaling according to the type of damages habitat and resources.[212]

[210] For a discussion on these parameters, see Thur, above n. 197, at 384 with many references.

[211] See, for instance, E.P. English, C.H. Peterson & C.M. Voss, ‘Ecology and Economics of Compensatory Restoration’, NOAA Coastal Response Research Center, CRRC (2009). The document is based on the division between ecological and human services. It describes concepts and methods of restoration scaling, including HEA, REA and survey-based valuations. However, the goal is not only to describe the state of the art, but mainly to provide a synthesis of techniques for specific habitats and resources.

[212] See English et al., above n. 211, for a complete study of HEA techniques based on the type of injured ecosystem.

As already said, HEA is then mainly applied to vessel groundings and restoration of sea grass beds and coral reefs, which are relatively small incidents. Based on monitoring studies, NOAA developed assessment and restoration planning protocols to quantify injuries, to project recovery and restoration times. For marsh habitats, which are the most often impacted by oil spills, trustees tend to choose a single or few metrics linked to lost primary eco logical services.

If replacements with habitats of the same type, quality and comparable value is not feasible or cost-effective, trustees make habitat tradeoff decisions and they often make more cost-effective use of restoration funds by replacing low-productivity with high-value habitats. Generally speaking, 'there is no one best-approach, but the aggregation approach should be tailored to the particular conditions of each case'.[213] Moreover, assessments are usually cooperative and trustees work together with responsible parties to value data and develop 'consensus-based' parameters for the HEA. Lastly, restoration plans are subject to public review which is an alternative and more cost-effective source of public preferences rather than survey-based valuations.[214] According to the US DOI, consensus-based approaches between trustees and liable parties, clear guidelines on scaling techniques and close coordination among various public authorities proved to be the best strategy to achieve restoration more quickly, more efficiently and more effectively, instead of monetary damages. [215]

3.4 The practice of compensatory restoration in the EU

While the equivalency analysis methods (HEA) for damage and remediation assessment have been employed in the United States for more than two decades under various statutory laws, it has never been applied in the EU until the ELD introduced it. It was explicitly incorporated in the EU Directive on environmental liability for environmental damage. However, unlike other non-market valuation methods, the use of equivalency analysis methods in the EU did not go along with a deep level of academic and legal scrutiny.

The Report on the 'Use of Resource Equivalency Methods in Environmental Damage Assessment in the EU With Respect to the Habitats, Wild Birds and EIA Directives'[216] within

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[213] Jones and DiPinto, above n. 207, at 343.

[214] Several scholars have criticized the HEA in the US because it does not assess correctly the benefits of restoration projects and money may end up in projects for which there is limited demand. Yet, according to the US DOI, the fact that public participation is required at various stages of the assessment of restoration plans is more efficient than a time-consuming collection of information on benefits that would ultimately be paid by the polluter. See Jones and DiPinto, above n. 207, at 345 and 347.

[215] Jones and DiPinto, above n. 207, at 345.

[216] J. Cox, 'Use of Resource Equivalency Methods in Environmental Damage Assessment in the EU With Respect to the Habitats, Wild Birds and EIA Directives', Deliverable D6B, REMEDE (2007).

the REMEDE project[217] revealed that the use of resource equivalency approaches among ecologists in the EU seems to be almost unknown but principles and approaches from the US are mainly used within the compensation and mitigation framework associated with the EU Birds, Habitats and Environmental Impact Assessment Directives.

'At this stage, no prior experience could be found of the use of resource equivalency methods in identifying compensatory strategies for neutralising accidental environmental damage'.[218]

However, within the REMEDE project, various case studies were considered as potential applications of the new Toolkit principles developed to estimate sufficient amounts of compensatory restoration.[219] It might be worth providing an overview of at least one of these cases (the chemical spill in Helsingborg, Sweden) to illustrate the existing state of knowledge in the EU. The spill occurred on 4 February 2005 when a chemical tank in Kemira near the municipality of Helsingborg collapsed releasing more than 16 thousand tons of toxic acid into the harbour connected to the Baltic Sea. The release had lethal effects on sea organisms, animals, sea plants and sediments, affecting up to 12ha and a depth of 10m.[220] The spill was followed by a compensatory restoration project with a cost of €100,000 that could provide 1 discounted ha of sea grass habitat services. Within REMEDE, Cole and Kriström tried instead to determine the total amount of (interim) environmental damage (debit) by using four different quantification metrics of habitat services (richness, abundance, biomass of invertebrates, habitat quality). The resulting total interim loss for the society was measured in biophysical terms: 33.1 discounted service hectares during the 4-year period needed to achieve full restoration. Then, three potential remediation options were identified and the gain (credit) in habitat services from one of this remediation project (planting of seagrass beds) was calculated. After discounting and obtaining the present value of gains and credits of habitat services, the appropriate amount of remediation was scaled to off-set the damage. Finally, the cost of the remediation option, including seagrass planting, re-project planning, administrative and permitting process and long-term monitoring costs, was calculated as indication of the magnitude of environmental liability associated with

[217] The REMEDE project has been designed to support the implementation of Annex II of the Directive 2004/35/EC on Environmental Liability with regard to the prevention and remedying of environmental damage (ELD). It was established within the 6th Framework Programme of the European Commission with the aim of testing and disseminating methods for determining the scale of remedial measures appropriate to offset accidental environmental damages. It draws on experience and methodological issues from the US and on the experience of the EU. It does not tackle the threshold of significant damage under the ELD, the estimation of how much primary remediation is needed and the best baseline to consider. It is only focused on resource equivalency analysis for compensatory remediation.

[218] Ibid., at 1.

[219] The cases are accessible at: http://web.archive.org/web/20140222151043/http://www.envliability.eu/docs/D12CaseStudies/D12CaseStudies.html [accessed 20 November 2023].

[220] S. Cole and B. Kriström, 'Tank Collapse and Chemical Release (Helsingborg, Sweden). Case study report', Deliverable D12, REMEDE, at 1 (2008).

the oil spill and based on a resource equivalency approach (€100,000).[221] Yet, this estimation was based on a previous US study and it was unsure whether costs would remain the same in the specific location of the Helsingborg harbour. Later, it was discovered that the 100,000-remediation project did not achieve the equivalency of lost and restored habitat services.[222]

Due to the scarce practice, it has been argued that remediation options in the EU should take in full account the lost services for human welfare.[223] Yet, as stressed by some EU experts: 'acceptable application of equivalency analyses requires technical knowledge (e.g., natural sciences, economics, law), data, stakeholder engagement, and sometimes a lengthy and costly negotiation process'.[224] Currently, the ELD does not provide public authorities with the needed capacity and technical resources for a correct application of equivalency analyses. A consensus-based approach to restoration should be needed, but there is no incentive to settle, nor any incentives to public participation in the design of restoration plans, as it happens in the US.[225] Moreover, the HEA does not encourage a dialogue among public authorities and groups with interests.[226] Lastly, there is no obligation to monitor the effects of restoration in the long term.[227]

4. Conclusions

This chapter analysed the remedies for environmental liability in the US and the EU in order to assess whether they are likely to achieve both the economic (optimal deterrence) and ecological (restoration) goal of liability laws.

In the US, polluters are exposed to a well-defined list of removal costs, interim losses pending recovery and costs of assessment.[228] Moreover, the environmental damage assessment often

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[221] Ibid., at 35.

[222] S. Cole, 'Environmental Compensation Using the REMEDE Toolkit: How Much Is Enough?', Stockholm presentation, REMEDE, 2008.

[223] E. Brans, 'Legal Analysis. Resource Equivalency Methods for Assessing Environmental Damage in the EU', REMEDE Report (2006).

[224] Lipton et al., above n. 120.

[225] This may bring to restoration not in line with the public demande: 'Une compensation hors site qui pourrait ne pas répondre à la demande de la population victime du préjudice n'est pas toujours satisfaisante'. See A. Bas, P. Gastineau, J. Hay & H. Levrel, 'Méthodes d'Équivalence et Compensation du Dommage Environnemental, 123 *Revue d'Économie Politique* 127, at 154 (2013). See also L. Krämer, 'The EU And The System of Environmental Loss and Damage. Liability, Restoration and Compensation', in B. Pozzo & V. Giacometti, *Environmental Loss and Damage in a Comparative Law Perspective* (2021), at 3ss, emphasising that there is not even incentive for the public (NGOs and individuals) to ask public authorities to intervene if they do not restore the injured environment in the absence of liable parties. In the US, NL and some other countries, if national authorities do not comply with the public request to intervene, people can go to court. This is not allowed under the ELD. 'Overall, the Directive does not offer effective means to protect the environment against illegal activities and hold the wrongdoer liable' (ibid., at 12).

[226] Bas et al., above, at 152.

[227] Ibid., at 153. The authors suggest the creation of a public authority in charge of valuing the effects of compensatory measures in cooperation with local and regional authorities.

[228] 33 USC 2706(d)(1). Additionally, OPA provisions enable separate claims for private losses to real property, profits, earning capacity, public losses to revenues and other costs.

relies on the HEA and it follows guidelines based on thirty years of experience.[229] In addition, a number of factors facilitate the fast attainment of restoration, which in turn minimises environmental costs. They are: clear responsibilities and timeframe for swift response actions, incentives for voluntary clean-up, incentives for settlement, clear obligations to use the money on restoration and creation of trustee councils, procedures of damage assessment based on the magnitude of the injury (A-type and B-type), consensus-based HEA and public participation in the design of restoration plans.

However, compensation of interim losses in the US is limited to ‘committed’ uses, standards for the estimation of non-use values were never adopted and the benefits of restoration projects are not accurately valued.

In the EU, polluters are exposed to the costs of primary, complementary and compensatory restoration, plus assessment costs. Like in the US, the restoration-based compensation has become the most common remedy because it is easier (based on fewer economic valuation methodologies), it ensures that the environment is returned to the baseline and it is flexible (one may opt for compensatory restoration even if primary restoration is feasible).[230]

However, a number of issues might reduce the likelihood to achieve both optimal deterrence and cost-effective restoration: recoverable costs are not very well-defined, there is no clear distinction between removal costs and restoration costs, public authorities are not obliged either to do clean-up or to recover removal and restoration costs, there are no precise timeframes and guidelines on restoration and HEA, there are no incentives to settle and the public is not involved in restoration plans. Moreover, compensatory restoration of interim losses and permanent losses cannot ensure full internalization due to the lack of expertise on HEA and ecological data on ecosystem services.

In conclusion, there are several reasons to doubt that the EU law on environmental liability (compared to the US) can fully internalise the environmental costs of accidents and achieve optimal deterrence in addition to cost-effective restoration.

[229] HEA is also used in many countries to determine the amount of compensatory mitigation needed to approve economic projects (ex ante) with future negative impacts on the environment. For additional references on this point, see W.D. Shaw & M. Wlodarz, ‘Ecosystems, Ecological Restoration, and Economics: Does Habitat or Resource Equivalency Analysis Mean Other Economic Valuation Methods Are Not Needed?’, 42(5) *Ambio* 628 (2013), at 630-631.

[230] On the lack of a hierarchy of remedies in the ELD, see G.M. van den Broek, ‘Environmental Liability And Nature Protection Areas Will The EU Environmental Liability Directive Actually Lead To The Restoration Of Damaged Natural Resources?’, 5(1) *Utrecht Law Review* 117, at 127 (2009).

Take-aways from chapter V and bridge to chapter VI

- Both in the US and the EU restoration has become the most common remedy because it is considered easier and cheaper and it ensures that the environment is returned to previous conditions.
- In the US, polluters are liable for a well-defined list of removal costs, interim losses pending recovery and costs of assessment.
- The environmental damage assessment in the US often relies on the HEA and it follows guidelines based on thirty years of experience.
- In the EU, a number of issues might reduce the likelihood to achieve both optimal deterrence and cost-effective restoration.
- (On compensatory restoration) neither the wording of the ELD is detailed and precise as in the US nor the economic valuation is made mandatory.
- The US law on Natural Resource Damage Assessment seems to be better placed to achieve cost-effective restoration and optimal deterrence compared to the EU Directive on Environmental Liability that was modeled after the former.

CHAPTER VI

Environmental Damage and Oil Spills

This chapter examines the extent to which polluters in real cases are exposed to the full costs of accidents, with a particular focus on the compensation of environmental damage beyond clean-up and restoration costs. This includes the non-use values of nature and interim losses. The analysis concentrated on large marine oil spills, given the availability of more data and scholarship on this topic. Additionally, it is widely acknowledged that international conventions governing oil spills do not typically expose polluters to liability for non-use losses and interim losses of nature. It is therefore of interest to examine the role of national liability laws in addition to the international legal framework for the compensation of oil spill damages. In the previous chapters, the question of whether the law is providing optimal incentives to achieve both restoration and deterrence in an efficient manner was addressed. This chapter goes even further by identifying additional aspects that equally matter for optimal deterrence and cost-effective restoration. These include optimal decisions on clean-up, incentives to claim compensation for 'pure environmental damage', length of liability lawsuits, post-accident monitoring and financial guarantees.

1. Introduction

The aim of this chapter is to assess to what extent polluters in real cases were liable for the environmental damage beyond clean-up and restoration costs, which includes non-use values of nature and interim losses. The analysis focuses on marine oil spills for which more scholarship and data are available compared to other kinds of accidents. Moreover, international conventions governing oil spills generally do not expose polluters to liability for non-use losses and interim losses of nature. Therefore, it is interesting to examine the role played by national liability laws in addition to the international legal framework for the compensation of the full social costs of accidents. Lastly, by comparing maritime oil spills on opposite sides of the Atlantic Ocean it is possible to infer additional conclusions on the cultural background underlying environmental liability laws. This chapter is therefore structured in three main parts.

The first part introduces the phenomenon of oil spills around the globe and it illustrates possible taxonomies of their consequences from both an economic and an ecological perspective.
The second part provides an overview of four marine accidents with huge ecological impacts (Amoco Cadiz, Exxon Valdez, ERIKA and Prestige) and it puts in the spotlight four aspects that seem to play a crucial role for the minimisation of the social costs of accidents:

- timing of clean-up and environmental costs of cleaning;
- claims of compensation for 'pure environmental damage' in addition to clean-up;
- number and length of civil liability lawsuits to claim environmental damage;
- further consequences of accidents likely to improve deterrence.

The third and last part summarises and compares the main characteristics of the cases considered and it draws normative conclusions further complemented by an anthropological view on the underlying cultural differences.
If the previous chapters tried to answer the question of whether the law is providing optimal incentives to achieve both restoration and deterrence in an efficient manner, this chapter goes even further identifying the additional factors emerging from the practice that may trigger or inhibit both a full internalization of environmental costs and cost-effective restoration. Indeed, aspects like the environmental costs of clean-up and the length of liability lawsuits may influence the efficient attainment of both deterrence and full restoration, whereas claims for environmental damage beyond clean-up costs and further consequences of oil spills can induce better deterrence. This analysis allows to depict a more realistic picture of nature valuation in liability lawsuits and to provide better recommendations on how to improve it.

2. Oil spills around the globe

Accidental oil spills happen when oil is transported from production sources to consumption locations due to the high potential of risks entailed in transportation. Soaring oil use for energy production and gas-to-oil switch considerably boosted the global demand of oil,[1] hence sharpening the likelihood of accidents throughout the world.[2] Yet, based on data gathered by the International Tanker Owners Pollution Federation Limited (ITOPF), both the number of oil spills and the total amount of oil spilled decreased worldwide over forty years from 1974 until

[1] Oil consumption increased by more than 54% in 36 years from 1973 to 2009 (IEA, 2011). For updated data on oil production, transport and demand, see IEA (2022), Oil Market Report - August 2022, IEA, Paris https://www.iea.org/reports/oil-market-report-august-2022[accessed 20 November 2023].
[2] D. Jin and H. Kite-Powel, 'Environmental Liability, Marine Insurance And An Optimal Risk Sharing Strategy For Marine Oil Transport', 10 *Marine Resource Economics* 1 (1995).

2010 (ITOPF 2010).[3] This is not contradicting evidence when compared with the growing oil demand during the same decades. It seems, indeed, that a few huge accidents were responsible for the most of oil spilled, such as the Hebei Spirit and the Prestige that accounted around thirty-five per cent of the total amount of oil spilled (212,000 tons). Figure 1 below illustrates the volume of oil spilt over the past five decades. The red part of the bars refers to the (oil spilt by) largest incidents for particular years (Atlantic Empress in 1979, Castillo de Bellver in 1983, etc.). Nineteen of the twenty largest oil spills since the 1970s occurred before the year 2000. Sanchi is the only major spill from 2010 until 2020 and it was responsible for about 70% of the quantity of oil spilt during the past decade. Based on the ITOPF statistics, the downward trend in oil spills continued in the decade from 2010 until 2020 despite the overall increase in oil trading. While in the 2000s there were 181 spills of more than 7 tons, a total of 196,000 tons of oil lost and 75% of the total spilt in just 10 incidents, in the 2010s there were 63 spills of more than 7 tons, a total of 164,000 tons of oil lost and 91% of this amount spilt in just 10 incidents.[4] As a general trend, in the 2010s it was recorded a ninety-five per cent reduction from the 1970s in terms of total oil spilt.

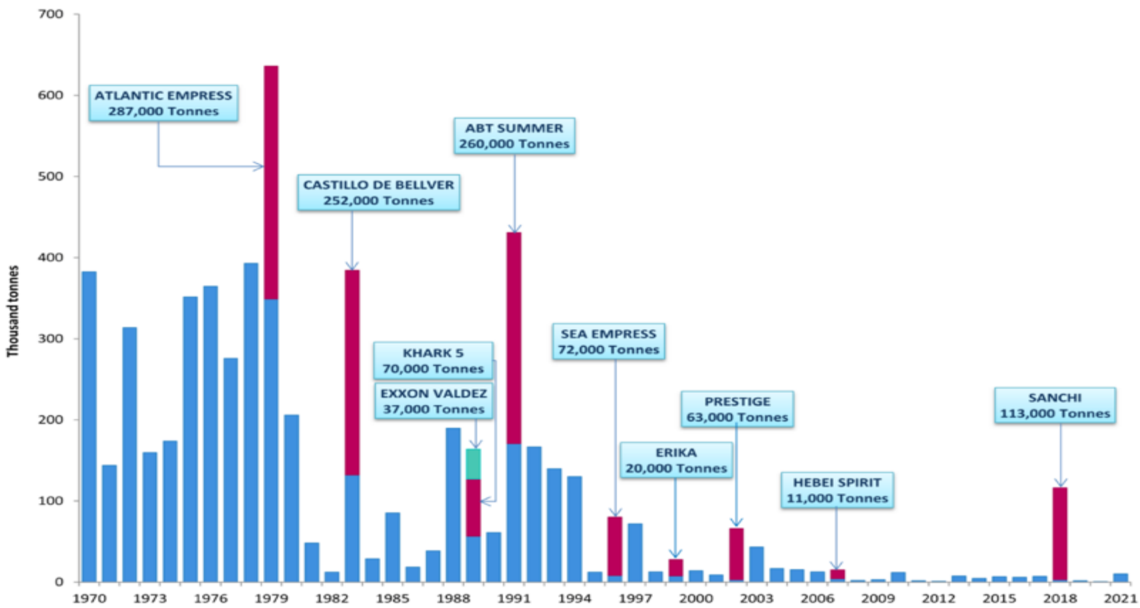

Figure 1 [Number of medium (<700 tonnes) and large (>700 tonnes) tanker spills, 1970-2022] [5]

[3] The International Tanker Owners Pollution Federation Limited (ITOPF), 2010. Historical data Statistics. Years 1974-2010. Available at: http://www.itopf.com/information-services/ data-and-statistics/ [accessed 20 November 2023].
[4] Ibid.
[5] The source is available online at: https://www.itopf.org/knowledge-resources/data-statistics/statistics/ [accessed 20 November 2023]

3. The consequences of oil spills

Notwithstanding the optimistic trend of the latest two decades, prevention of oil spills remains of paramount importance due to the severity of damage caused by oil spills. It is indeed well known that major impacts are brought to the society, the environment and the economy of the affected areas (fishing, tourism and other sectors of the economy). These effects have been reported in several impact assessment studies following catastrophic marine accidents.[6] In 2014, Chang and others carried out a large literature review of more than 300 academic, industry, governmental papers and reports on the consequences of oil spills.[7] The most prominent accidents in the review include: the 1978 Amoco Cadiz oil spill offshore of Brittany (France), the 1989 Exxon Valdez oil spill in Prince William Sound (Alaska, US), the 1996 Sea Empress spill in the UK, the 1999 ERIKA oil spill off the coast of Brittany, the 2002 Prestige oil spill offshore of Spain and Portugal, the 2007 MT Hebei Spirit oil spill in South Korea and the 2010 BP Deepwater Horizon oil spill in the US. The study highlights both the main factors which are likely to influence the effects of oil spills and it is also a precious source of information on the consequences of oil spills.

Regarding the factors influencing the final effects, one of the main findings in Chang is that the magnitude of consequences depends not only on the oil spillage itself but also on the marine physical environment and the management response. The factors related to the oil spillage itself that can influence and dramatically increase the amount of damage may be represented by the ship safety features, the location, the closeness to shores and the spillage rate. For instance, Kontovas et al. (2010) reported that the number of oil spills over 7 tonnes since the 1970s dramatically decreased with the introduction of new mandatory regulations aimed at making ships safer[8] and Nyman (2012) estimated that shoreline clean-up is 4 or 5 times more expensive

[6] For the Amoco Cadiz spill, see T.A. Grigalunas, R.C. Anderson, G.M. Brown, R. Congar, N.F. Meade & P.E. Sorensen, 'Estimating the Cost of Oil Spills: Lessons from the Amoco Cadiz Incident', 2(3) *Marine Resource Economics* 239 (1986). For the Exxon Valdez, see R.T. Carson & S.M. Walsh, 'Preventing Damage from Major Oil Spills: Lessons from the Exxon Valdez', 32(3-4) *Oceanis: Serie de Documents Oceanograthiques* 351, at 354 (2006). For the ERIKA oil spill, see F. Bonnieux & P. Rainelli, 'Learning from the Amoco Cadiz Oil Spill: Damage Valuation and Court's Ruling', 7(3) *Industrial & Environmental Crisis Quarterly* 169 (1993).

[7] S. Chang, J. Stone, K. Demes and M. Piscitelli-Doshkov, 'Consequences Of Oil Spills: A Review And Framework For Informing Planning', 19(2) *Ecology and Society* 1 (2014).

[8] C.A. Kontovas, H.N. Psaraftis and N.P. Ventikos, 'An Empirical Analysis Of IOPCF Oil Spill Cost Data', 60(9) *Marine Pollution Bulletin* 1455 (2010). But see Alló and Loureiro (2013) who conducted the first worldwide meta-damage analysis of oil spills aimed at assessing the effects of the legislation on the magnitude of damage caused by oil spills and they found that, on average, the application of strict liability reduces the damages of spills by $ 241.43 million, so confirming what was previously anticipated by Shavell in 1984 on the joint use of liability and regulations for deterrence (S. Shavell, 'A Model Of The Optimal Use Of Liability And Safety Regulation', 15(2) *The RAND Journal of Economics* 271 (1984). Likewise, Alberini and Austin found that strict liability for clean-up costs raises the level of care of potential polluters. See: M. Alló and M.L. Loureiro, 'Estimating

than off-shore clean-up.[9] Alló and Loureiro (2013) found that a 1% increase in oil spilled raised the level of damages by US$ 0.718 million.[10] Loureiro et al. (2006) estimated that spills releasing oil slowly and continuously tend to accrue long-term costs due to the need of multiple response efforts.[11] Additionally, specific aspects of the marine physical environment are likely to influence the impact of the spill and especially the areas exposed to oil. They relate to weather conditions, tides and currents. Law and Kelly (2004) described that wind speeds above 30 knots prevented at-sea recovery operations for much of the initial stages of the spill during the Sea Empress incident at the entrance to Milford Haven in 1996.[12] However, the effects of the spill were milder than expected (in light of the quantity of oil spilt) for many reasons, including the fact that most of the fish and crabs were already offshore due to the season. Carls et al. (2001) found that tides and currents influence the natural energy available to natural oil dispersal and the effectiveness of chemical dispersal.[13] Lastly, the management response can also influence the magnitude of damage based on the technologies employed and the governance approach. Apparently, pre-spill contingency plans can ensure more rapid and effective response strategies.[14] Loureiro et al. (2005) also showed the role of volunteers and military staff (local capacity) for fast clean-up and an overall reduction in cleaning expenses. As to the technologies for cleaning, all procedures seem to further damage the marine environment or, at least, slow down the natural recovery of the area (see §4). This has been early stated in the ecological literature[15] and further proved with global data by Sell et al. (1995).[16] To sum up, the consequences of oil spills are generally dynamic and depending on

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A Meta-Damage Regression Model For Large Accidental Oil Spills' 86 *Ecological Economics* 167 (2013); A. Alberini and D. Austin, 'Accidents Waiting To Happen: Liability Policy And Toxic Pollution Releases', 84(4) *The Review of Economics and Statistics* 729 (2002).

[9] T. Nyman, 'Evaluation of methods to estimate the consequence costs of an oil spill', *SKEMA Seventh Framework Programme* (2009). Kontovas *et al.,* above n.8, also estimated that shoreline clean-up might cost up to US$ 29,000/ton, while off-shore clean-up might cost US$ 300,00/ton.

[10] This was in line with a previous study by White. See: I.C. White, 'Factors Affecting The Cost Of Oil Spills' *International Tanker Oil Pollution Fund* (2002). Available at: http://www.itopf.com/_assets/costs02.pdf [accessed 20 November 2023]

[11] M.L. Loureiro, A. Ribas, E. López and E. Ojea, 'Estimated Costs And Admissible Claims Linked To The Prestige Oil Spill', 59(1) *Ecological Economics* 48 (2006).

[12] R.J. Law and C.A. Kelly, 'The Impact Of The "Sea Empress" oil spill', 17 *Aquatic Living Resources* 389 (2004).

[13] The terms of natural oil dispersal and chemical dispersal will be clarified in the following sections. On the evidence about the effect of tides on oil dispersal, see: M.G. Carls, M.M. Babcock, P.M. Harris, G.V. Irvine, J.A. Cusick, and S. D. Rice, 'Persistence Of Oiling In Mussel Beds After The Exxon Valdez Oil Spill' 51 *Marine Environmental Research* 167 (2001).

[14] I.C. White and F.C. Molloy, 'Factors That Determine The Cost Of Oil Spills' *2005 International Oil Spill Conference, IOSC 2005*, 10470 (2005).

[15] M.S. Foster, J.A. Tarpley, and S.L. Dean, 'To Clean or Not to Clean: The Rationale, Methods, and Consequences of Removing Oil from Temperate Shores' 6 *The Northwest Environmental Journal* 105 (1990).

[16] D. Sell, L. Conway, T. Clark, G.B. Picken, J.M. Baker, G.M. Dünnet, A.D. Mclntyre and R.B. Clark, 'Scientific Criteria to Optimize Oil Spill Clean-up' Proceedings of the 1995 International Oil Spill Conference, American Petroleum Institute, Washington D.C. 595 (1995).

both natural conditions and the human effort in returning the environment to the status quo ante.

The next crucial issue is thus to clarify what the consequences of oil spills are. In order to do that, it might be useful to showcase the main frameworks proposed by the economists that worked on the environmental damage assessment of the major oil spills that will be analysed in this chapter.

Standing from a purely economic perspective, it must be recalled that a monetary quantification of natural resources is possible to the extent that there is a measurable change of the demand related to the damaged environmental goods and services. If a demand can be identified, it means that the environmental services are part of the utility function of consumers and their value can be assessed from the perspective of the welfare theory.[17]

Having said that, a first easy-to-read framework of consequences is the one proposed by Bonnieux and Rainelli in 1993 and in 2005, drawing respectively on the Amoco Cadiz and the ERIKA oil spills.[18] According to the two economists, it is possible to distinguish a category of direct costs of oil spills, which includes costs of clean-up and restoration, from a category of indirect costs,[19] which includes the 'physical adverse effects of the spill, in spite of clean-up efforts',[20] i.e. costs to marine resources, to tourism, to recreation and amenities, and to biodiversity. Figure 2 below illustrates this distinction:

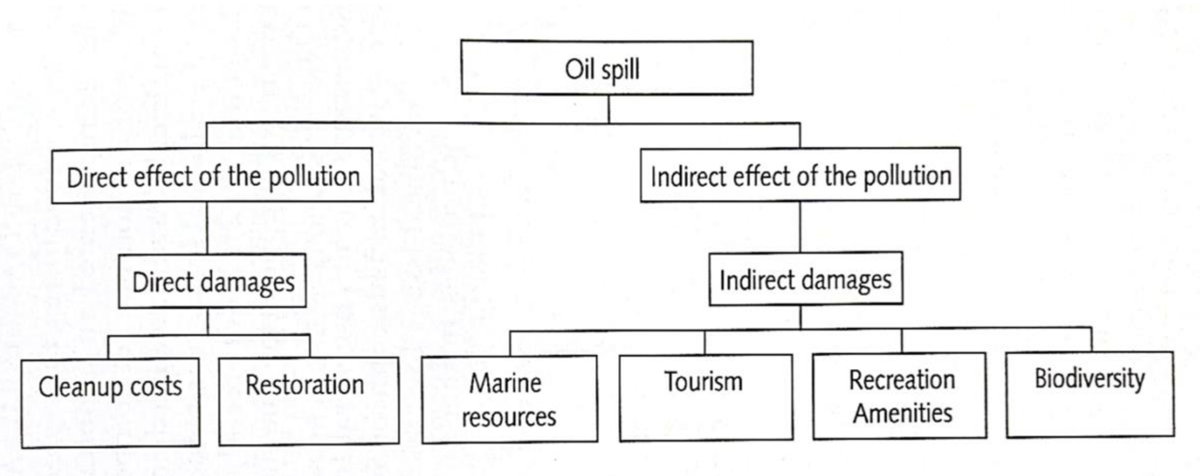

Figure 2 [Linkage between the ERIKA oil spill and the social costs of the spill] [21]

[17] F. Bonnieux and P. Rainelli, 'Economic Analysis Of The Consequences Of Pollution By ERIKA : Problems And Methods' [Analyse économique des conséquences de la pollution par l'ERIKA : problématique et méthodes], Post-Print hal-02833681, HAL, at 165 (2005).

[18] Ibid.

[19] On the utilitarian view of natural resources underlying this definition of indirect environmental damages, ibid. at 158.

[20] F. Bonnieux, 'Economic Assessment of Market and Non-market Damages of Oil Spills', 32(3/4) *Océanis* 321, at 323 (2006).

[21] Ibid, at 324. The figure does not show health damages because their magnitude was insignificant both in the ERIKA and in the Amoco Cadiz (ibid., at 323).

More precisely, the amount of indirect damages is the reduction in the service flow, having the status quo ante as reference. These services include those with commercial value (fishing, recreational activities)[22] and those related to non-commercial values, such as the loss of amenities of local people or the loss of biodiversity to the extent that it affects the social welfare.[23] In order to have a monetary quantification, Bonnieux and Rainelli (2005) listed all services possibly provided by the shoreline based on the well-known distinction between use and non-use values.[24]

The services that may be related to use values are:

- industrial transportation (port),
- residences,
- tourism,
- fishing,
- sailing,
- rowing boating,
- bathing,
- recreational fishing,
- hiking,
- walking.

Regarding the method of valuation, all services can be assessed through market prices or the prices of other goods and services that are traded to get the benefit of a non-traded environmental good (sailing and rowing boating, bathing and recreational fishing, hiking and walking).

The only service lacking use value is flora and fauna existence. Indeed, it is a common view that the mere existence of flora and fauna provides services that are intrinsically different from the amenities derived from landscape contemplation or living in a beautiful place, which inevitably include the use of environmental services. Given that flora and fauna existence has only non-use value, it has to be assessed directly with the contingency valuation methodology.

[22] Like many other environmental accidents, the ERIKA spill affected ecosystems that supported economic activities (tourism, urbanised areas, maritime activities) that depend on the quality of the environment.

[23] From a welfare theory perspective, the increase or reduction of services provided by the ecosystems is a possible way to express the change of utility derived from natural resources. See Bonnieux and Rainelli (2005), above n. 17, at 158.

[24] See chapter II.

Whatever the method of environmental damage assessment is and the category of costs, the geographical level and the concerned population have to be identified for accurate estimates. This is relevant not only for services with market prices (e.g., seafood)[25] but also for services that are not traded in the market. The final estimate in fact varies if it is based on the whole population or just on some residents. Lastly, the two categories of direct and indirect damages should not be regarded as independent. While the environment recovers at a natural rate, human (clean-up and restoration) activities may influence such dynamics by accelerating or conversely slowing down the return of the damaged environment to previous conditions. If they determine a diminution of indirect damages, then it is possible to say that direct damages measure this environmental gain.[26] The figure below simplifies the inverse relationship between direct and indirect damages. It shows the flow of services over time. Immediately after the accident, all types of services (indirect costs) decline and they gradually return to the baseline with different time periods: very short (path 1) with high restoration, medium time (path 2) with low restoration and very long (path 3) with natural recovery. Yet, it is very difficult to trade-off direct and indirect damages (infra §4).

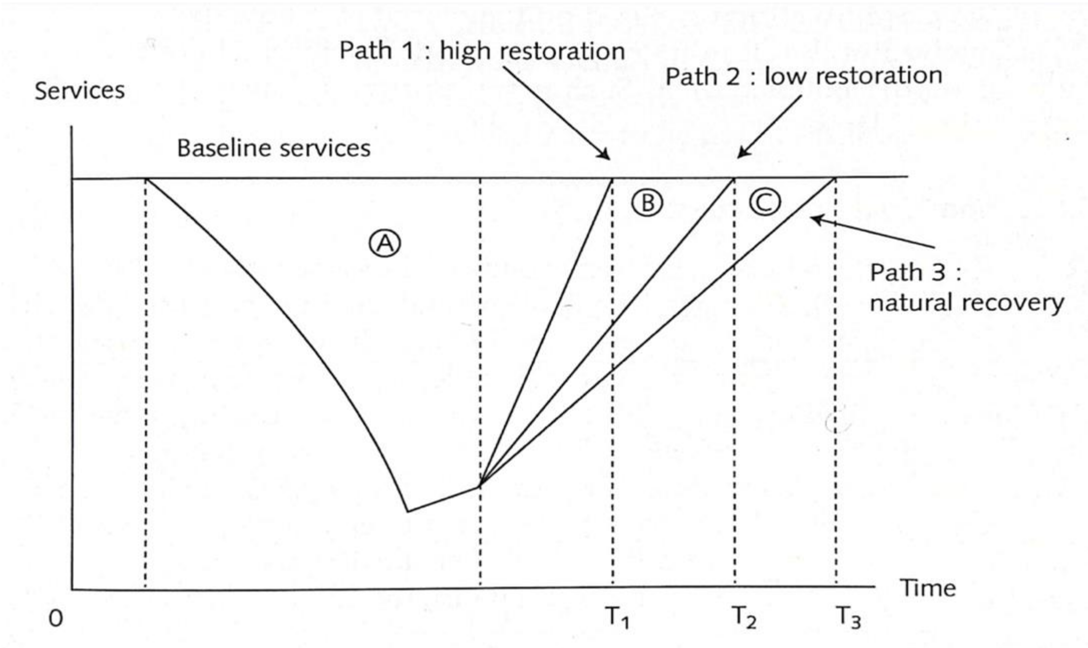

Figure 3 [Dynamics of the damage to the environment] [27]

[25] In this case, to estimate indirect damages it is possible to compare the consumer surplus before and after the accident, assuming that the market before the occurrence of the accident was in a condition of perfect competition with maximum producers' and consumers' surplus.
[26] However, it is also possible that clean-up has negative environmental impacts (infra, §4).
[27] Bonnieux and Rainelli (2005), above n. 17, at 161-162.

Hay and Thébaud (2006)[28] proposed a more detailed categorisation of the consequences of oil spills by resorting to four categories:

1. Changes in bans and regulations introduced in response to the pollution in order to restrict the use, for instance, of shorelines and natural resources.
2. Changes in consumption and production habits of economic agents in response to the perception that pollution has affected the quality of products.
3. Changes in the consumer and/or producer surplus due to direct and indirect modifications in the quantity of natural resources available.
4. Mitigation measures (clean-up and restoration) adopted to limit the extent of damages.

The figure below illustrates these four categories and it emphasises that oil spills' consequences can be on a) markets; b) regulations; c) ecosystems, while the 'environmental damage' in a narrow sense (pure environmental damage) regards only the latter category (ecosystems) when there is an impact on non-use values and indirect use values that here refer to the use of the ecosystem by the same ecosystem.

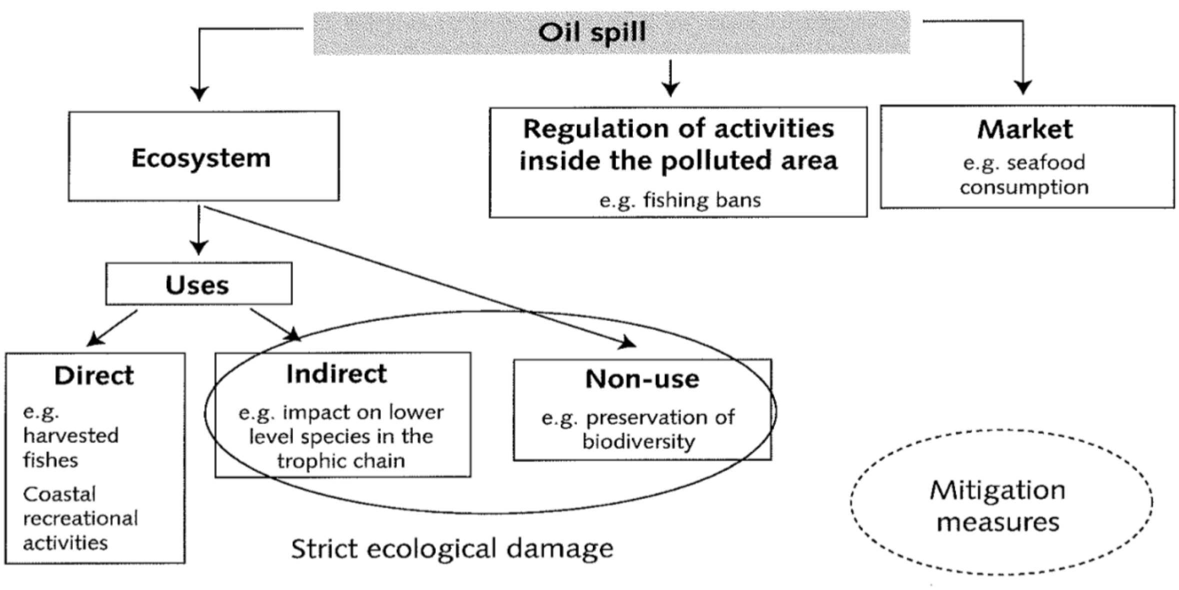

Figure 4 [Types of impacts of oil spills on economic systems][29]

[28] J. Hay and O. Thébaud, 'Including Ecological Damage in the Monetary Valuation of Oil Spill Impacts: An Assessment of Current Practice', 32(3/4) *Océanis* 297, at 301-302 (2006).

[29] Hay and Thébaud, above n. 28, at 303. The authors pointed out a further category of damages related to the consequences of pollution on private and public goods, such as boats, sea infrastructures and cars that might require reparation after accidents. Yet, this category is not mentioned in their framework since it represents only a limited amount compared to the total costs of oil spills (at 303, footnote 6).

Another framework that needs to be mentioned was provided by Carson and Walsh (2006)[30] and it was followed in the United States since the Exxon Valdez spill (while the previous frameworks have been used in the French oil spills). The Carson & Walsh schema starts from the question of 'who' has been harmed and it then looks at 'how much'. The first category is thus given by those who have been directly harmed, i.e. fishermen (lost profits). The second category includes who have been less directly impacted but that nevertheless suffered from a reduction in business activities due to the oil spill, i.e. hotel operators (lost profits). The third category concerns the tourists who cancelled their hotel reservations and that had to stay at home (lost consumer surplus rather than lost profits). Lastly, if the accident occurred in a protected area a fourth category of losses comes in the picture: passive-use (or non-use) losses[31] for the members of the public who care about the protected areas (lost consumer surplus).[32] Carson and Walsh explained that the traditional approach followed in the US prior to the Exxon Valdez oil spill was only focused on 'direct' and possible 'indirect' losses (lost profits). However in response to the Exxon Valdez, the US government moved to a broader assessment scheme to make the shippers of oil pay the 'full compensation' and take appropriate precautionary measures.[33]

Lastly, to complement the economic perspective, it might be interesting to see how the impacts of oil spills may be categorised in the ecological literature and, particularly, in Chang et al. (2014).

[30] R.T. Carson & S.M. Walsh, 'Preventing Damage from Major Oil Spills: Lessons from the Exxon Valdez', 32(3-4) *Oceanis: Serie de Documents Oceanograthiques* 351, at 354 (2006).

[31] The term 'passive use values' was first coined by the US DC Court of Appeals to include what economists have usually called existence values and option values. Carson explains that passive use values are simply a specific category of pure public goods that do not suffer from congestion externalities, meaning that their use does not diminish the value of the natural resources (ibid., at 355).

[32] R.T. Carson, N.E. Flores and R.C. Mitchell, 'The Theory And Measurement Of Passive-Use Value', in I.J. Bateman and K.G. Willis (eds), *Valuing Environmental Preferences: Theory and Practice of the Contingent Valuation Method in the US, EU, and Developing Countries,* at 97 (1999).

[33] In the words of Carson and Walsh: 'Members of the public who care about the protected area also suffer losses in consumer surplus. The number of such agents, however is potentially very large and involvement of the government is now clearly unavoidable if "full" compensation is to be paid'. (…) The economic consequences of using only lost profits from direct impacts should be clear: the economic cost to the shipper of oil will be less than the economic damages caused by the spill. As a consequence, oil shippers will not invest enough in preventing the damage from spills and there will be more damage from oil spills than is socially desirable' (Carson and Walsh, above n. 30, at 355).

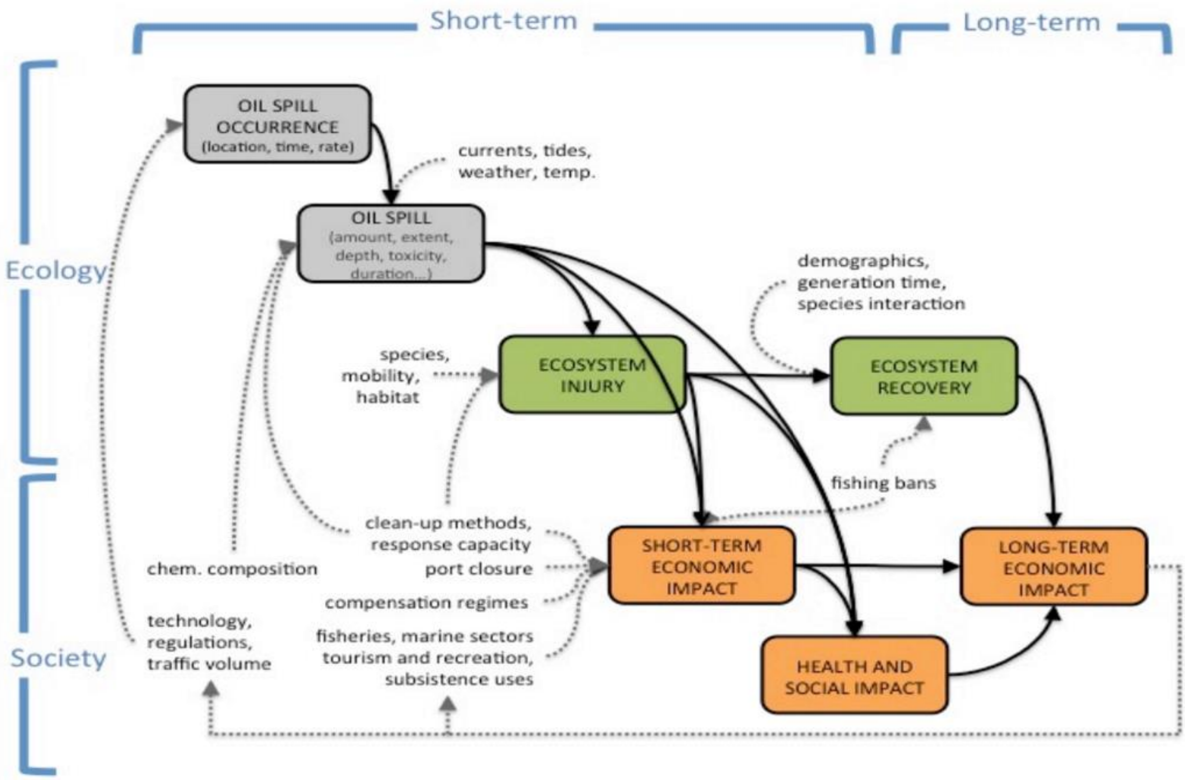

Figure 5 [Impacts of oil spills][34]

In the figure above, the boxes represent all the consequences of oil spills (grey boxes for oil spill outcomes, green boxes for ecosystem consequences and orange boxes for social consequences).[35] Moreover, the solid lines connect oil spill occurrence and impacts with socioeconomic impacts, the dotted lines link exogenous variables with outcomes and, lastly, all the terms out of the boxes are variables. This framework is useful to understand that from an ecological perspective the first level of consequences relate to the oil spills in themselves, the second level to the ecosystem responses and the third to the consequences on human society. People can be eventually affected by oil spills in three ways:[36]

- direct human health damages, i.e. by breathing oil vapours;
- indirect human health damages, i.e. by eating seafood with accumulated toxins;
- direct economic losses, i.e. to fishermen or owners of recreational services.

[34] Chang et al. (2014), above n. 7, at 27.
[35] In ecology the consequences of oil spills on the environment are considered separately from the social consequences: while the former refer to changes in the stare of the environment the latter relate to changes in the conditions of human beings. Clearly, the distinction is not based on the monetary or non-monetary nature of the effects.
[36] This useful framework is used by Chang et al. (2014) drawing on the previous paper by Webler and Lord. See: T. Webler and F. Lord, 'Planning For The Human Dimensions Of Oil Spills And Spill Response', 45 *Environmental Management* 723 (2010).

This is an alternative way to look at the consequences of oil spills which differs from the previous economic frameworks because it prioritises the impacts on ecosystems over the impacts on society. Moreover, public health impacts are more explicitly mentioned.

Summarising all these frameworks, oil spills may possibly lead to the following four types of costs:

- costs of clean-up and restoration (direct costs, mitigation costs);
- costs to the environment (indirect costs, losses of non-use values);
- financial losses (indirect costs, costs to the market);
- human health damage.

Set aside financial losses and human health damage that do not fall into the scope of this dissertation, the first two categories will be put in the spotlight while looking at specific cases. Interestingly, non-use values have been included in all the economic frameworks, as:

- flora and fauna existence (Bonnieux and Rainelli);
- lost value for the same ecosystem, 'strict environmental damage' (Hay and Thébaud);
- lost consumer surplus for protected areas (Carson and Walsh).

Now, before jumping into the cases, it is worthwhile to touch upon a crucial and often overlooked issue, i.e. how clean-up techniques may affect the magnitude of environmental consequences. This is indeed crucial in view of assessing the general efficiency of remedies and their likelihood of fully internalising environmental damage. Seemingly, a possible interplay between clean-up techniques and the environment suggests that optimal remedies should be such that the polluter (or who may be in charge of clean-up) is correctly incentivised to minimise not only clean-up costs but also the environmental costs caused by the latter. Therefore, the next two sections will review the scientific literature explaining how the consequences of oil spills can evolve without treatment or clean-up (i.e., with natural recovery) (§4) and how they can evolve when clean-up operations interact with natural recovery (§5). Understanding these processes lays the basis for the subsequent case analysis.

4. Post-spill evolution without treatments: natural recovery

Oil spilled in the marine environment following an accident largely degrades over time through natural processes, like evaporation, mixing, precipitation, photooxidation (sunlight) and bacteria.[37] Some studies show that these kinds of processes, especially bacteria and wave actions, contribute to disperse oil in a more effective way than human interventions.[38] For this reason, scientists argue in favour of natural recovery if clean-up is likely to cause high environmental impacts.[39]

After the start of the physical and chemical degradation, a process of natural recolonisation of coastal communities begins at some point.[40] It means that plants and animals which are characteristic of the local ecosystems are re-established in the damaged environment. This natural recovery is a 'process' evolving through three stages of changing composition in population size and age structure.

The first stage is the initial colonization of the affected area by the so-called 'macroscopic opportunists' (green seaweeds on rocky shores or algae within a salt marsh).[41] The second stage is the recovery or progress towards (but not yet attaining) the establishment of a natural biota. Finally, the 'recovered' stage when the natural biota is within the range of dominance, diversity, abundance and zonation expected for that habitat. Figure 6 on the next page summarises the above-mentioned stages.

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[37] A.R.G. Price, 'Impact Of The 1991 Gulf War On The Coastal Environment And Ecosystems: Current Status And Future Prospects' 24(1) *Environment International* 91 (1998).

[38] Gutierrez studied the role of bacteria for the degradation of oil after the Deepwater Horizon oil accident. He inferred that bacteria played a critical role for degrading oil in deep waters since less chemical pollutants were found on the water surface. See T. Gutierrez, 'Identifying Polycyclic Aromatic Hydrocarbon-Degrading Bacteria In Oil-Contaminated Surface Waters At *Deepwater Horizon* By Cultivation, Stable Isotope Probing And Pyrosequencing', 10 *Review in Environmental Science and Bio/Technology* 301 (2011).

[39] Foster et al. (1990), above n. 15.

[40] This is in line with Figure 3 above by Bonnieux and Rainelli.

[41] This has been reported straight after the Amoco Cadiz spill and the Exxon Valdez spill. See Lehre (1984) who compiled data and figures to report on the fate of oil and the recovery of the biota after the Amoco Cadiz oil spill of 1978 in Brittany (France) and Southward (1979) who studied the process of recolonization of rocky shores in West Cornwell following the Torrey Canyon oil spill in 1967. See K. Lehre, 'The Amoco Cadiz Oil Spill- At a Glance', 15(6) *Marine Pollution Bulletin* 218 (1984); A.J. Southward 'Cyclic fluctuations in population density during 11 years recolonisation of rocky shores in West Cornwall following the Torrey Canyon oil spill in 1967', in E. Naylor and R.G. Hartnoll (eds), *Cyclic phenomena in marine plants and animals: Proceedings of the 13th European Marine Biology Symposium, Isle of Man, 27 September - 4 October 1978,* Oxford, Pergamon Press, 1979.

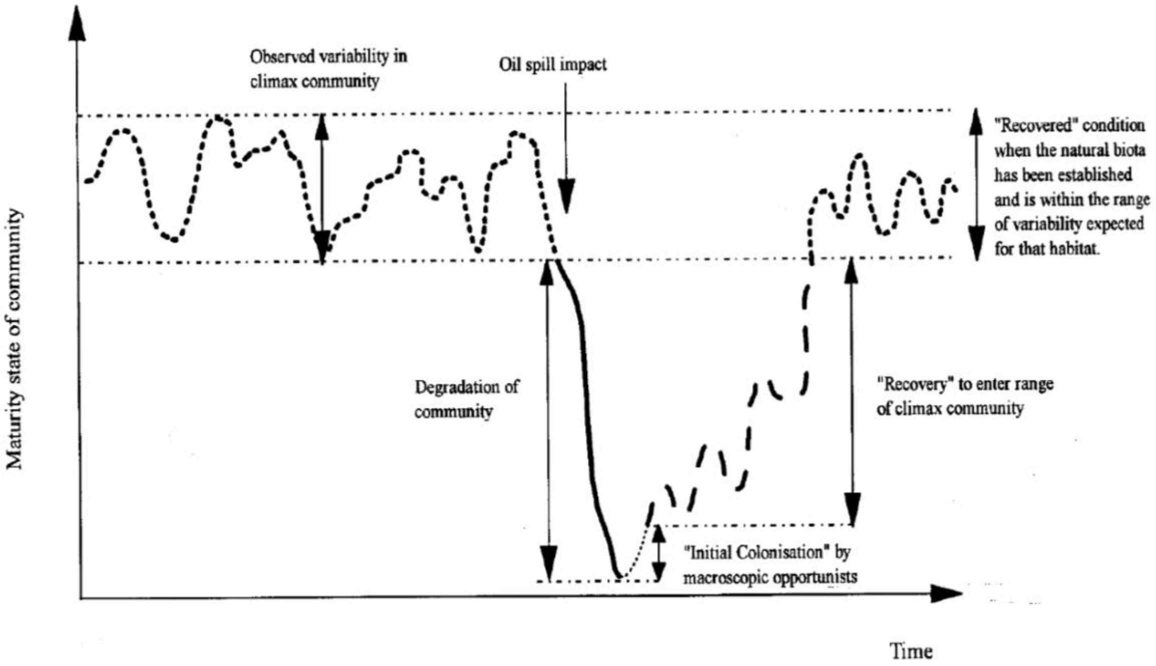

Figure 6 [Impact of an oil spill on an intertidal community][42]

The dotted line shows the observed variability of communities before oil spills which in some cases is very large.[43] The sharp dip in the line indicates that the community is degraded or destroyed after an oil spill impact. Clearly, there is no single end point of recovery. That makes hard to establish when the recovered ecosystem returned to the precise condition before the accident. Notwithstanding such uncertainty, natural recovery allows to considerably save clean-up costs, including the same costs of cleaning up the environment, especially with vulnerable ecosystems that are more sensitive to traditional treatment methods (see §5). This does not mean that natural recovery should always be preferred to clean-up. It rather means that if the goal of liability laws is the minimisation of accidents' social costs, the costs and benefits of clean-up have to take into account also the environmental long-term costs of cleaning compared to natural recovery. Studies show indeed that natural recovery aims at the full restoration of the environment, while chemical treatments aim at returning only certain services to the pre-existing conditions before the accident (see §5.2). The next section will better clarify this point.

[42] Source of the figure: Sell et al., above n. 16, at 596.

[43] The state of communities is naturally fluctuating in terms of dominance, diversity, abundance. Great variability also emerges during recovery and in the recovered stage. On this point: J.M. Baker, R.B. Clark, P.F. Kingston & R.H. Jenkins, 'Natural Recovery Of Cold-Water Marine Environments After An Oil Spill', 13th Annual Arctic Marine Oil Spill Program Technical Seminar (1990).

5. Post-spill evolution with treatments: clean-up

Despite the process of natural recovery, oil spills are often followed by clean-up, which can be classified into two main categories: conventional and biological. [44]

Conventional *in situ*[45] methods are:

- mechanical, i.e. containment techniques that use barriers to enclose floating oil and prevent it from spreading (e.g., 'booms', 'skimmers', adsorbent materials);[46]
- thermal, i.e. burning techniques (*in situ* burning) that burn the oil with minimal equipment;
- chemical, i.e. techniques that change the physical and chemical properties of the oil through the employment of dispersals and solidifiers.

Booms and skimmers do not change the physical and chemical properties of the oil, so it can be reused. Yet, they rely on specific conditions to be effective (calm waters, a certain gravity and surface tension). Adsorbent materials complement booms and skimmers by cleaning the remaining oil. Sorbents can be natural organic, natural inorganic or synthetic. The latter are the most widely used for their capacity to absorb 70 – 100 times their weight (natural organic can absorb 3 – 15 times their weight). Adsorbent materials have the disadvantage that they cannot be collected after spreading on the water, so they sink and sediment. Synthetic materials are also not biodegradable.

Thermal techniques and namely in situ burning is the most successful and effective technique after oil spills, especially with refined oil products that burn quickly. Yet, they might cause long term negative effects on plants, marine animals and human health.

Chemical techniques are the best for the protection of sensitive marine habitats and the shoreline because they rapidly make the oil less sticky so it does not stick on sea birds and vegetation. Dispersants are applied by spraying the water from vessels and aircrafts, therefore they are suitable to rough seas, as well. High winds enhance their mixing with the water. The problem is that they have inflammable nature and can be highly toxic for human and ecological health (see §4.2).

[44] P.E. Ndimele et al., 'Remediation of Crude Oil Spill', in Ndimele (ed), *The Political Ecology of Oil and Gas Activities in the Nigerian Aquatic Ecosystem*, at 370ss (2017).

[45] *In situ* remediation works by decontaminating the water and/or soil where the accident occurred.

[46] For instance, dams were built in the Doñana case (see chapter V) and 'curtain booms' (subsurface skirts) were used in the BP oil spill (see chapter III).

Convention *ex situ*[47] methods are instead:

- soil excavation or dredging, i.e. removal of contaminated soil to an off-site location for burying or burning;
- incineration, i.e. removal of contaminated soil and transport to off-site facilities to burn the oil and reduce the toxic substances;
- soil washing, i.e. removal of contaminated soil and treatment with water and chemicals to separate contaminated sand from uncontaminated soil;
- thermal desorption or soil roasting, i.e. treatment of contaminated soil with hot temperature to enhance vaporisation and separation of contaminants;
- pump and treat, where water is pumped out of the ground and treated in surface water facilities to remove contaminants and to reuse the water.

All these methods are quite resource-intensive, they require high operational costs, space and workers. They can also cause additional exposure to hazardous substances by extraction. Nevertheless, pump and treat is very common.

Based on the above, the fastest remediation techniques are also the most intense: use of chemical dispersants to break down the oil into small droplets, *in situ* burning so that oil is combusted, mechanical clearance of large areas, extensive removal of the top surface of the substratum, high-pressure hot water hoses to wash off the shoreline and rocky shores.

Conversely, moderate techniques, such as low-pressure flushing, limited cutting of vegetation, low-toxicity dispersants, manual cleaning, require much more time (but they are also cheaper).

5.1. Optimal clean-up and uncertainty

As it can be inferred from the above, decisions on clean-up operations and, namely, on their time, duration and resource intensity, should be based on an overall assessment of various factors, including the ecological state of the environment, its sensitivity, its economic and ecological importance in addition to the costs and the logistics of clean-up.[48] However, since

[47] *Ex situ* remediation works by removing the contaminated soil or water to clean them up in another location.

[48] Pasquet and Denis classified clean-up techniques employed in the Amoco Cadiz oil spill based on the following criteria: productivity (the quantity of wastes collected per unit of time), impact on ecosystems (very productive equipment can be detrimental to the ecosystem), selectivity (quantity of oil collected in the wastes), effectiveness, accessibility to the coast, availability, cost, compatibility with handling and disposal of waste. For instance, manual clean-up is very selective but not so productive and the only one available in inaccessible areas. It also ensures fast natural recovery thanks to the minimal human disturbance caused to ecosystems. The only way to increase productivity is by mechanizing clean-up with the use of equipment but this might come at the expenses of the environment. R. Pasquet and J. Denis, 'New Developments In Beach Clean-Up Techniques', *2005 International Oil Spill Conference, IOSC 2005*, 5135 (2005).

the decision-making also needs to be quite fast to raise the likelihood of success of the remedial action,[49] decisions in the end turn out to be based on incomplete information.

Reasoning from a theoretically economic perspective, the desired level of cleanliness can be identified with a cost effectiveness analysis. More specifically, the optimal level of clean-up is where the marginal clean-up cost matches the marginal improvement in the quality of the shoreline (diminution of ecological damage).[50] Considering that clean-up efforts normally have a declining productivity, the equilibrium point happens where the sloping curve of clean-up costs crosses the rising curve of environmental improvement:

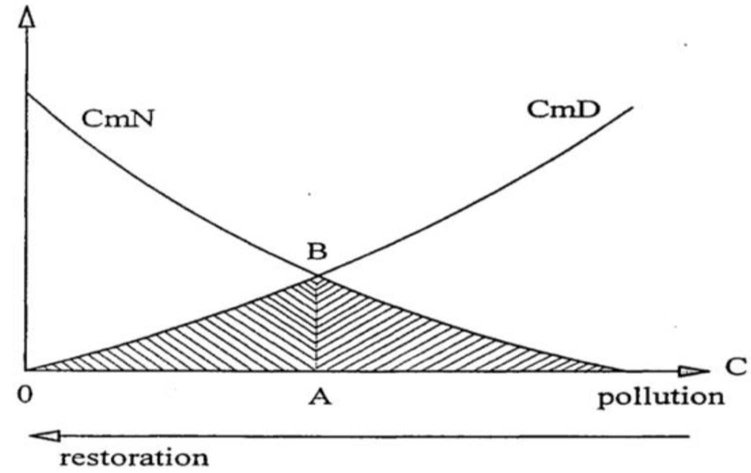

Figure 7 [Optimal level of clean-up] [51]

In Figure 7 above, the CmN curve refers to clean-up costs per metric ton of oil spilled. They are very high when pollution is close to 0 (the cost of removing an additional ton of oil gets extremely high when the shoreline is almost clean) and they tend to be very low for large amounts of oil spilled on the right side (the cost of removing an additional ton of oil is lower if the environment is heavily polluted). The CmD refers instead to the marginal benefit of clean-up, i.e. the services provided by the healed natural resources. Moving from the right to the left, the more you clean the less the pollution But the proportion between benefits and costs tends to diminish while getting closer to a pollution level equal to zero. At point B the quantity AC

[49] While for some world's coasts, contingency plans taking into account their vulnerabilities have been already issued and they allow decision-makers to implement clean-up in a faster manner, for the majority of coasts these plans do not exist and decisions on interventions after oil spills need to be adopted in very short timeframes (ibid., at 596).

[50] F. Bonnieux & P. Rainelli, ´Learning From The Amoco Cadiz Oil Spill: Damage Valuation And Court's Ruling` 7(3) *Industrial & Environmental Crisis Quarterly* 169, at 174 (1993).

[51] Ibid., at 174-175. The authors base their analysis on clean-up costs following the Amoco Cadiz oil spill.

of oil has been removed, while the quantity A0 remains and the total cost spent in clean-up is ABC. As a consequence, optimal compensable damage should theoretically include clean-up costs (AC) plus the remaining damage (A0). However, as pointed out by Bonnieux and Rainelli (1993, 2005), this theoretical approach cannot work in practice for various reasons.

First, it is static while natural resources are dynamic.

Secondly, it requires specific information on weather, tides, wave action, geomorphology and action of the dispersants available. Often, information on weather conditions and the state of ecosystems is incomplete when deciding upon an emergency response because some pieces of scientific information are unavailable and/or the needed information is costly.

Therefore, it is almost impossible to determine ex ante the optimal level of clean-up expenses and it is very likely that expenses for clean-up will turn out excessive or useless or even harmful for the environment. The reasonableness and the relevance of the measures taken can be discussed in the light of the ex post results.[52] Often, the only possibility to get closer to the optimal level is by monitoring clean-up operations.[53] However, when it comes to the environmental impacts of clean-up, the state of the art in science allows to make some predictions already ex ante.

5.2 The environmental impacts of clean-up

The costs of clean-up are not only of financial nature and related to the people. In addition to these, ecologists often stress the possible negative impacts of clean-up on the ecosystem to be cleaned.[54] Although it is true that oil spills often cause severe ecological damage, the cleaning of rocky shores and beaches can considerably increase negative impacts on ecosystems, turning short-term ecological damage in long-term one.[55] The reasons for that are readily summarised below.

[52] See also Bonnieux and Rainelli (2005), above n. 17, at 158: 'From this point of view, the concept of optimal clean-up level is hard to define ex ante considering the uncertain evolution of ecosystems and the uncertain climate scenarios' (En ce sense, la notion de niveau optimal de nettoyage s'avère ex ante difficile à définir compte tenu des incertitudes qui pèsent sur l'évolution des processus qui régissent le fonctionnement des écosystèmes touchés et des aléas climatiques).

[53] Ibid., at 161.

[54] Much has been written on the effects of oil spills and remedial treatments, particularly in the domain of shore ecology. See Sell, D. et al. (1995), above n. 16, at footnotes 6,7,8 and 36.

[55] D. Broman, B. Ganning and C. Lindblad, 'Effects Of High Pressure, Hot Water Shore Cleaning After Oil Spills On Shore Ecosystems In The Northern Baltic Proper' 10(3) *Marine Environmental Research* 173 (1983); R. de la Huz, M. Lastra, J. Junoy, C. Castellanos and J.M. Viéitez, 'Biological Impacts Of Oil Pollution And Cleaning In The Intertidal Zone Of Exposed Sandy Beaches: Preliminary Study Of The Prestige Oil Spill' 65(1-2) *Estuarine Coastal and Shelf Science* 19 (2005).

First of all, chemical dispersants used to break down oil droplets might be extremely toxic.[56] Moreover, the combination between chemical dispersants and oil might be even more harmful for marine species.[57] Also, dispersal can increase the bioavailability of oil and further expose organisms to the hydrocarbon-dispersant compound, which might ultimately enter the food chain.[58] An additional downside of clean-up is that it is likely to slow down the process of natural recovery which aims at the full restoration of the ecosystem. Chemical treatments aim instead at returning only certain services to the pre-existing conditions before the accident.

The interaction between clea–up and natural recovery has been in the spotlight of scientists for many years. The first global in-depth study was carried out in the 1990s by the American Petroleum Institute.[59] The goal was to review the worldwide literature on the effects of oil spills (including the American Trader and the Exxon Valdez) and find out correlations with the recovery process of both rocky shores and salt marshes. Within the study, data on oil spills were classified according to: shore type, exposure, oiling and treatment in order to give a unified framework for comparison. Based on these characteristics, the timing and duration of recovery processes were compared with and without clean-up treatments. The study found out that in untreated shores (without clean-up):

[56] R.S. Judson, M.T. Martin, D.M. Reif, K.A. Houck, T.B. Knudsen, D.M. Rotroff, M. Xia, S. Sakamuru, R. Huang, P. Shinn, C.P. Austin, R.J. Kavlock and D.J. Dix, 'Analysis Of Eight Oil Spill Dispersants Using Rapid, In Vitro Tests For Endocrine And Other Biological Activity' 44(15) *Environmental science & technology* 5979 (2010); K. Sriram, G.X. Lin, A.M. Jefferson, W.T. Goldsmith, M. Jackson, W. McKinney, D.G. Frazer, V.A. Robinson and V. Castranova, 'Neurotoxicity Following Acute Inhalation Exposure To The Oil Dispersant' 74(21) *Journal of Toxicology and Environmental Health, Part A* 1405 (2011). Bejarano in a recent review found that hazard concentration for fifty-four dispersants fell between the moderate and slightly toxic range but the toxicity of chemical dispersants basically depends on application and dilution. See: A.C. Bejarano, 'Critical Review And Analysis Of Aquatic Toxicity Data On Oil Spill Dispersants' 37(12) *Environmental Toxicology and Chemistry* 2989 (2018). A previous literature review on oil spill dispersants by Fingas (2014) pointed out that the first motivation for using dispersants is to reduce the impact of oil on shorelines, the second is to reduce the impact on birds and mammals on the water surface and the third is to trigger the biodegradation of oil in the water column. It seems that the debate around these benefits never ended. Dispersant effectiveness remains a major issue with many more factors potentially undermining it (amount of dispersants, oil composition, sea energy, water temperature and salinity). Likewise, benefits on wildlife remain largely unknown and many papers argue that the current dispersant formulations may inhibit oil biodegradation or at least the effects on biodegradation seem to be controversial. M. Fingas, S. Science and E. Alberta, 'A Review of Literature Related to Oil Spill Solidifiers 1990-2008 – for Prince William Sound Regional Citizens' Advisory Council (PWSRCA C), Anchorage, Alaska (2008).

[57] Some algae develop protective mechanisms against crude oil pollution but it is unknown whether marine mammals can do the same (M.F. Wolfe, G.J.B. Schwartz, S. Singaram, E.E. Mielbrecht, R.S. Tjeerdema, M.L. Sowby, 'Influence of dispersants on the bioavailability and trophic transfer of phenanthrene to algae and rotifers', 48(1) *Aquatic Toxicology* 13 (2000)). See also: A. Cohen, D. Nugegoda and M.M. Gagnon, 'Metabolic Responses Of Fish Following Exposure To Two Different Oil Spill Remediation Technique' 48(3) *Ecotoxicology and environmental safety* 36 (2001); M.Z. Vosyliene, N. Kazlauskiene and K. Joksas, 'Toxic effects of crude oil combined with oil cleaner simple green on yolk-sac larvae and adult rainbow trout Oncorhynchus mykiss', 12(3) *Environmental science and pollution research international* 136 (2005).

[58] A. Mascarelli, 'Deepwater Horizon: After The Oil', 467(7311) *Nature* 22 (2010). Yet, it is unsure whether less aggressive clean-up measures (e.g., skimmers) can avoid major negative effects on ecosystems.

[59] Sell et al. (1995), above n. 16, at 601. It may be argued that the American Petroleum Institute had a specific interest in this study because it led to the conclusion that it would be better to spend less (or nothing) on clean-up operations. However, the fact that the same conclusions are supported by several scientific papers suggests that the long-term environmental impact of fast and intense clean-up treatments on vulnerable ecosystems (i.e., rocky shores and beaches) should be taken into account for optimal decisions on clean-up.

i) initial colonization generally occurred within 6 months and it was completed within 12 months;

ii) recovery tended to start within 12 months and it seemed to be largely completed by 24 months;

iii) the natural process of recovery in the absence of oiling or clean-up was almost done within three years after clearance.

Conversely, clean-up operations retarded the start of initial recovery of heavily oiled rocky shores by an average of four months (and 11 months in the case of the Exxon Valdez spill),[60] especially in the case of exceptionally intense clean-up methods. Data on polluted salt marshes confirm this trend, more exactly treatments may speed up the start of the recovery stage but they tend to cause a longer time of full recovery (e.g., for the Amoco Cadiz).[61]

In conclusion, the study proved the existence of a correlation between clean-up and longer recovery time of the natural biota, especially for vulnerable resources. The scientific explanation is that cleaning operations, especially intense treatments and toxic dispersants, sterilize the substratum, kill any biota that survived the initial effects of the oil spill and remove or diminish physic-chemical characteristics useful for the development of plants and animals.[62] Given the marginal or negative impact of clean-up operations on natural recovery, from an ecological perspective there is very little scientific justification for clean-up operations after many oil spills, especially where they are affecting vulnerable ecosystems. Ecological recovery is expected to naturally occur within three or five years (respectively, for rocky shores and salt marshes). In view of minimising not only clean-up costs, but also the ecological costs of oil spills, it should be thus crucial to choose the proper clean-up technique by weighting their economic benefits against the negative ecological impacts and obtain the "net environmental benefit" of clean-up treatments.[63] From this point of view, intensive clean-up might be advisable only if the persistence of oil on the shoreline poses a worse threat to the environment than the adverse effects of clean-up itself.[64] Alternatively, biological methods (bioremediation) based on the use of microorganisms to degrade naturally the pollutants might be a cheap and eco-friendly solution. Yet, biodegradation relies on the availability of nutrients, the

[60] Ibid.

[61] Ibid.

[62] D. Kirchman and R. Mitchell, 'A Biochemical Mechanism for Marine Biofouling', *OCEANS 81* 537 (1981).
This is even more probable in high-energy ecosystems where natural recovery would be extremely rapid and non-intervention would be more recommended (Sell above n. 16).

[63] API-American Petroleum Institute/NOAA-National Oceanic and Atmospheric Administration, 'Inland Oil Spills: Options For Minimizing Environmental Impacts Of Freshwater Spill Response' (1994).

[64] API-American Petroleum Institute and MBC Applied Environmental Sciences (Costa Mesa, Calif.), 'Oil Spill Response: Options For Minimizing Adverse Ecological Impacts' (1985).

concentration of oil, the time and the extent of natural recovery already achieved.[65] The other limitation is that sector specific best practices on sustainable remediation are still on their way for being produced and incorporated in the law.[66]

The whole discourse above boils down to the initial consideration (see taxonomies of social costs in §3) that environmental accidents can cause both environmental costs that are monetarily quantifiable (clean-up and restoration) and 'pure environmental costs' (Hay and Thébaud) beyond use values. The polluter should thus receive optimal incentives to minimise both categories of environmental costs. Do liability laws incentivise the optimal internalisation of pure environmental costs of clean-up on vulnerable ecosystems? Likewise, do they incentivise the optimal internalisation of interim losses in case of late clean-up?

5.3 Private interests and overcleaning

Apparently, decisions on clean-up are often driven merely by 'socio-economic factors', such as recreational, touristic, commercial activities and visual amenities (use values)[67] or 'public and political pressures', to say that in the words of Chang.[68] Also other scholars emphasised that the amount of clean-up expenses is considerably influenced by 'local expectations for what is clean and the relative costs of clean-up in the local context'.[69]

Boudouresque et al. (2019)[70] brought up an interesting case of overcleaning on the coasts of Provence (France) following a minor oil spill. On 7 October 2018, the ro-ro[71] ferry Ulisse released some 600 m3 of fuel after colliding with a containership 28 km north-west of the Cape Corse (Corsica). Due to strong winds and currents, the oil reached the coasts of Provence and the Port-Cros National Park (PNCP). Interestingly, the director of the Scientific Council of the

[65] Ndimele, above n. 44, at 376.

[66] A.O. Thomas, 'Application Of Sustainable Remediation Principles To Upstream Oil And Gas Projects Opportunities And Challenges'. Paper presented at the SPE African Health, Safety, Security, and Environment and Social Responsibility Conference and Exhibition, Maputo, Mozambique, September 2014. The author defines sustainable remediation as 'the practice of demonstrating, in terms of environmental, economic and social indicators, that the benefit of undertaking remediation is greater than its impact and that the optimum remediation solution is selected through the use of a balanced decision-making process'. The goal is to include in the decision-making also the costs for the environment, human health and the views of local stakeholders in order to maximise the benefits of local societies.

[67] Foster et al. (1990), above n. 15.

[68] Chang et al. (2014), above n. 7, at 28.

[69] K.W. Wirtz, N. Baumberger, S. Adam and X. Liu, 'Oil Spill Impact Minimization Under Uncertainty: Evaluating Contingency Simulations Of The Prestige Accident', 61(2-3) *Ecological Economics* 417 (2007). See also: Alló and Loureiro (2013), Fingas (2012), Kontovas et al. (2010) and Nyman (2009).

[70] C.F. Boudouresque, A. Blanfuné, G. Martin, M. Perret-Boudouresque, I. Taupier-Letage, 'The Virginia Oil Spill In Provence: A Tale Of Inappropriate Over-Cleaning', Rapp. Comm. int. Mer Médit., 42, 2019, p.100 ⟨hal-03065573⟩.

[71] "Ro-ro" stays for roll-on/roll-off. Ro-ro ships are cargo ships designed to carry wheeled cargo, such as cars, trucks, buses that are loaded with their own wheels or with the help of a platform vehicle. Ro-ro ships differ from lo-lo ships where cargo is loaded and unloaded with the help of a crane.

PNCP issued a warning against the potential risks for the ecosystems as a result of heavy and intense cleaning methods on rocky shores and beaches. It was particularly recommended not to use chemical dispersants and hot water high pressure, '*except for rocky shores accessible to pedestrians close to beaches*'.[72] Furthermore, it was recommended not to clean 'areas of high ecological sensitivity', to remove as little sand and wood as possible from the beach, to preserve Posidonia oceanica banquettes of dead leaves and to manually remove only oiled surface layers.[73] In the end, upon request of the French authority (PNCP), clean-up operations were conducted by a private company internationally recognized for its competence, that took also into account the recommendations of the ships' insurers. Overcleaning was avoided only in the core area of the Park, thanks to the warnings of the Scientific Council, but most of the area accessible to tourists (and also inaccessible) underwent intensive and disproportionate cleaning. In the view of the authors, decisions on clean-up were driven by the insurers' private interests to remove visible traces of oil and, thus, to 'limit claims for compensation for economic damage (loss of profit) and monitoring costs'.[74] The French case points out two further issues that add up to the previous analysis, making the picture much more vivid and yet complicated.

The first issue is the role of insurers in the post-spill phase and their possible conflicting interests when it comes to clean-up. Given the interplay between clean-up and environment (above §4.2), it is intuitive that if insurers are asked only to cover the economic losses caused by environmental accidents they will not take into account the environmental costs of clean-up techniques. It can be argued that the insurance coverage somehow influences the choice of clean-up measures if the latter is upon the insurers. The insurance coverage is in turn strictly linked to liability regimes and the lack of consideration for environmental losses unveils the inefficiency of environmental liability laws (if insurers would need to pay back also the environmental losses, they would need to include also the latter in their cost benefit analysis).

The second issue is the regulatory framework that might play a counterbalancing role when taking decisions on clean-up operations. The case above shows that environmental regulations may be effective to avoid the environmental impact of heavy cleaning methods on vulnerable environments, such as protected areas. This beneficial interplay between nature conservation laws and liability laws could be achieved in the case considered due to the existence of a public

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[72] Boudouresque et al. (2019), above n. 70.

[73] Ibid.

[74] Ibid., "The result is that a very minor oil spill, occurring 9 months before the next tourist season, was transformed by the polluters themselves (via the insurers), by carrying out a disproportionately large -scale clean-up operation, into a major ecological damage, the natural restoration of which will take at least 10 years.".

authority competent on the protection of natural resources (without conflicting interests) and the availability of ecological data proving the negative consequences of cleaning on specific ecosystems.[75]

6. From Costs to Coasts: a comparative analysis of costs in four major oil spills

There is another aspect that matters for the full internalization of the environmental costs of accidents and this is whether the 'pure environmental damage' (beyond clean-up and restoration) can be claimed and compensated under liability laws and whether this effectively happens.

In §3 the consequences of oil spills have been investigated from both an economic and an ecological perspective. One would therefore expect that these consequences correspond to a single global estimate or measurements. However, the social costs of oil spills in practice may lead to at least three categories of numbers, following the empirical analysis by Thébaud et al. (2004):[76]

- estimates of the damage based on academic studies applying economic valuation methods;
- claims for compensation before the Court;
- compensation eventually paid to the accident's victims or settled by the Court.

This classification results from evidence collected in major oil spills.

The table on the next page summarises the different numbers emerging from four major cases that will be later analysed in depth. Subcategories of costs have been converted into percentages in order to make comparisons.

[75] On the interplay between liability and nature conservation laws, see V. Fogleman, ´The Threshold for Liability for Ecological Damage in the EU: Mixing Environmental and Conservation Law´ in C.-H. Born, A. Cliquet, H. Schoukens, D. Misonne & G. Van Hoorick (Eds.), *The Habitats Directive in its EU Environmental Law Context: European Nature's Best Hope*? (2014), at 181.

[76] O. Thébaud, D. Bailly, J. Hay & J. Pérez-Agundéz, 'The Cost of Oil Pollution at Sea: An Analysis of the Process of Damage Valuation and Compensation Following Oil Spills', in A. Prada Blanco & X. Vasquez Rodriguez (coordinators), *Economic, social and environmental effects of the "Prestige" oil spill"*, International Scientific Seminar, Santiago de Compostela, March 7-8th, 2003, Consello da Cultura Galega.

				TOTAL COSTS				STRICT ENVIRONMENTAL COSTS		
Oil Spill	Country	Year	Oil spilled (tons)	Estimated ($ million)	Clean-up costs (% of total estimated costs)	Losses of profits (% of total estimated costs)	Losses of amenities (% of total estimated costs)	Estimated ($ million and % of total estimated costs)	Claimed ($ million and % of total estimated costs)	Paid to victims (P) or settled by Courts I ($ million) and % of total estimated costs
Amoco Cadiz[77]	FR	1978	223,000	1,161	36%	14%	40%	110.5 **10%**	119.5 **10%**	5.3 (P) **2%**
Exxon Valdez[78]	US	1989	37,000	11,859	29%	/	/	2,800 **4.3%**	2,800	**1,417** I **8.4%**
ERIKA [79]	FR	1999	20,000	[916-1,077]	33%	61%	N/A	[26.2 – 33.6] **6%**	**0**	**0** (P)
Prestige [80]	ES	2002	63,000	890	66%	31%	N/A	29 **3%**	**0**	**0** (P)

Table 1 [Breakdown of social costs of oil spills]

[77] The sources of the estimated costs of the Amoco Cadiz oil spill are: NOAA, 'Assessing the Social Cost of Oil Spills: the Amoco Cadiz Case Study', US Department of Commerce, National Oceanic and Atmospheric Administration (1983) and F. Bonnieux & P. Rainelli, *Catastrophe Écologique et Dommages Économiques. Problèmes d'Évaluation à Partir de l'Amoco-Cadiz*, INRA Éditions (1991). While for the claimed and granted monetary awards, we refer to J. Hay and O. Thébaud, 'Measuring The Costs Of Oil Pollution At Sea: An Analysis Of The Process Of Damage Valuation And Compensation Following The Amoco Cadiz Oil Spill', 4 *Économie Appliquée* 159 (2002).

[78] The source of the estimated costs of the Exxon Valdez oil spill is R.T. Carson, R.C. Mitchell et al., 'Contingent Valuation and Lost Passive Use: Damages from the Exxon Valdez Oil Spill', 25(3) *Environmental Resource Economics* 257 (2003), while the claimed amounts come from IOPC documents and the granted amounts from M. McCammon, 'Xestion da recuperación económica e ambiental: o Consello de Administradores Fiduciarios da marea negra do Exxon Valdez', in A.P. Blanco & X.V. Rodriguez, *Economic, Social and Environmental Effects of the Prestige Oil Spill*, Consello de Cultura Gallega, at 113 (2004).

[79] The source of the estimated costs of the ERIKA oil spill is the study by the French accountants Mazars and Guérard appointed by the association Ouest Littoral Solidaire. See Cabinet Mazars and Guérard, 'Évaluation Des Prejudices Économiques, Écologiques And Sociaux Suite Au Naufrage De l'ERIKA Sur Les Territoires Des Regions De Bretagne, Pays De La Loire Et Poitou-Charentes' (2001). IOPC documents provide instead the information about the claimed and the granted amounts.

[80] The source of the estimated costs of the Prestige oil spill Loureiro et al. (2006), while IOPC documents provide information about the claimed and the granted amounts.

Based on Table 1, within the total costs of oil spills:

- clean-up and response costs represent the most significant component of social costs of oil spills with an average share of half (between one third and two thirds of) the total costs;
- profit losses gain the second position with an average of 32% (between 4 and 61%);
- losses of amenities are a less considerable share of the total costs (between 19 and 40%);
- strict environmental damage accounts for the smallest share of total costs of oil spills (an average of 7%, between 0 and 21%).[81]

Focusing on the last subcategory, it has been reported that the environmental damage (intended in a narrow sense, as damage to non-use values and indirect use values) represents the less frequently assessed share of monetary damages under both US and international liability regimes within the estimated, claimed and granted damages.[82] However, it seems that the environmental damage (in a narrow sense) has been more frequently claimed under US laws rather than the international regime.[83] Moreover, data show that the share of environmental damage tends to change during the valuation and compensation process with a declining trend: only a part of the initial estimates is eventually included in claims and payments awarded. For instance, the share of environmental damage estimated in the Amoco Cadiz case corresponded to the claimed share, but less than the 2% of the estimated amount was eventually awarded. Unexpectedly, neither in the ERIKA oil spill nor in the Prestige one, damage strictly related to losses of non-use values of the environment was claimed and/or awarded despite it surely occurred. Conversely, in the Exxon Valdez case a high monetary award for environmental (and non-use values) losses was settled in Court. Why these differences? Which factors play a role for the calculation, the claim and the award of environmental costs in the post-spill aftermath? Hay and Thébaud (2006) identified four possible factors having negative impacts on the amounts granted for compensation of environmental damage:[84]

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[81] These percentages can be found in Hay and Thébaud, above n. 28, at 311. The authors adopt a very narrow notion of environmental damage that excludes direct use values (e.g., from recreational activities) and that only includes non-use values and indirect use values. From their perspective, impacts on direct use values occur through the modification of the abundance, the capacity to renew and the accessibility of natural resources. Indirect impacts on use values refer instead to modifications of ecological components that in turn affect the availability of natural resources. See Hay and Thébaud, above n. 28, at 302.

[82] See on this point the study by D. Helton and T. Penn, 'Putting Response and Natural Resource Damage Costs in Perspective', in *International Oil Spill Conference 1999* (1999). The study estimated that environmental damage have been estimated in less than one percent of the 5,000 to 10,000 oil spills yearly reported in the US coasts. However, these data refer to environmental damage in a broad sense, including also the impacts on direct use values (recreational). Likewise, the 2005 annual report of the IOPC Fund found that claims for environmental damage (stricto sensu) were mentioned only in 6% of the cases compensated under the IOPC Fund out of 136 oil spills occurred over the previous three decades. Given the supplementary role of the Fund (in addition to the CLC convention and intervening when victims cannot be fully compensated under the CLC convention).

[83] This conclusion is inferred by Hay and Thébaud, above n. 28, at 308-309.

[84] Hay and Thébaud, above n. 28, at 314.

1. lack of interest to raise a claim due to outweighing costs compared to benefits;
2. lack of a public agent legally entitled to claim compensation for environmental damage on behalf of the general public (this was specifically the case in the ERIKA and Prestige);
3. exclusion of environmental damage (losses of non-use values) from the heads of damage admitted by liability laws (this is the case of the international legal practice and the IOPC Fund convention that does not allow to claim compensation for non-monetary quantifiable losses);
4. length of compensation process (this was the case in the ERIKA spill where the Court rejected the claims for restoration costs on the grounds that natural recovery made those expenses unneeded at the time of the trial and exactly ten years after the accident; this is also the case of the international compensation regime that only allows costs of (reasonable) restoration measures).

Another factor to consider in view of the final internalization of environmental costs is whether the spill occurred on-shore or off-shore. If oil spills take place far from land, they tend to be less noticeable even if they last over long periods of time. Unfortunately, the location determines whether the accident would be both cleaned up and finally compensated.[85]

The factors listed above clearly extend the focus of this analysis beyond a mere comparison of valuation methods (see chapter II). They raise the need to investigate more in depth the specific conditions under which strict environmental damage has been estimated, claimed and eventually compensated in real cases. The analysis below therefore tries to fill in this gap by comparing four major oil spills in the US and the EU over the past five decades. For each case, the following sections will provide a general presentation followed by a description of the clean-up, of the environmental damage assessment and the remedies adopted in the end.

7. The Amoco Cadiz case (1978)

The Amoco Cadiz's oil spill of 1978 represents the first case in Europe attracting attention around the assessment of environmental damage for the high magnitude of the latter and given that the case was followed by fourteen years of litigation. During the morning if 16 March

[85] See M. Fourcade, 'Cents and Sensibility: Economic Valuation and the Nature of "Nature" ', 116(6) *American Journal of Sociology* 1721, at 1742 (2011) for examples of accidents that remained unnoticed and undetected for many years because of their location, sometimes even in spite of the local mobilization threatened by the leakage of oil.

1978, the supertanker Amoco Cadiz, carrying 228,000 tons of highly toxic oil[86] from Iran and Arabia and owned by the American Oil Company, ran aground opposite the small fishing port of Portsall in Brittany (northern French coast). The tank spilled around twice the amount of oil released in the Torrey Canyon of 1967 near the Coast of Wales[87] and around six times the amount of oil later spilled from the Exxon Valdez in 1989 in Alaska. The oil could be smelt up to 50 km away from the accident site and given the bad weather, it became impossible to intervene on the wreck in time. 'Within a fortnight, almost all of the cargo was spilled in the sea',[88] going to damage around 200 miles (around 300 km) of the French coast from the West to the East, driven by winds.[89] On the following day, the French Ministry of the Environment decided to launch a chemical and ecological study programme of the spill. The following impacts were counted: dozens of millions of invertebrates (260,000 tons) of all species dead (mollusks, sea urchins and other sea-bed organisms), 22,000 birds lost, oyster beds destroyed and some species like seals and puffins could not be recovered anymore.

7.1 The clean-up

The clean-up of the region was coordinated by the French government and implemented by the seaside towns from Brest to Saint-Brieuc[90] under the national Plan POLMAR (contingency plan for such accidents) and it was extremely time-consuming and resource-intensive: it lasted six months and involved the participation of the French army, thousands of soldiers, heavy machinery and volunteers from all over France. Upon experts' advise, the French Ministry banned the use of dispersants at a depth of less than 50 meters and less than 3 miles off the coast. The baseline was established by a team of ecologists (helped by the University of West Brittany students) by taking sediment samples from the coast not yet reached by pollution, by measuring the probability of finding dead birds along the coast (only one quarter was found) and trying to get reliable records of the number of the number of organisms dead. The low level of biodiversity in the area facilitated the creation of ecological records of reference that would

[86] On the composition of the oil, see L. Laubier, 'Compensating Ecological Damage: A Personal Experience', 32(3/4) *Océanis* 279, at 282 (2006). The toxicity is based on the specific size, density and water solvability of oil compounds. The type of oil transported by the Amoco Cadiz consisted of highly light molecules which are the most toxic, least viscous and most water soluble. That led to the oil spreading out in thinner and thinner slicks in the water column.
[87] Notably, the accident of the supertanker Torrey Canyon (123,000 tons of oil spilled) was the very first major spill before the Amoco Cadiz oil spill. For the first time, methods to clean-up and to tackle the extensive natural damage had to be discovered.
[88] Laubier, above n.86, at 282.
[89] Commission d'Enquête du Sénat, 'La Catastrophe de l'Amoco Cadiz', Rapport de la Commission d'Enquête du Sénat (1978).
[90] An agreement was needed between the Prime Minister and the local departments to entrust the cities to do the clean-up and then seek compensation, given that the accident occurred in public waters (the State had full sovereignty).

be much more difficult in the Mediterranean marine ecosystem.[91] Various techniques of clean-up were tried, with positive impact on the production of new scientific knowledge. The research on clean-up techniques was mainly funded by the Amoco owner and US federal agencies and it was almost twice the amount spent by the French State for the same purposes.[92] Indeed, two research programmes were funded on the assessment of the environmental damage, one by the National Oceanic and Atmospheric Administration (NOAA) and one by the European Commission (authors: Bonnieux and Rainelli). The costs of clean-up were born by the French government.[93]

7.2 The litigation and the environmental damage assessment

Since France subscribed to the International Convention on Civil Liability for Oil Pollution Damage of 1969 (implemented in France in 1975), in 1978 Amoco accepted to pay to a fund the maximum amount of liability allowed by the Convention at the time of that accident (77 million francs, equivalent to about $16 million at the 1978 rate of exchange). Yet, the fund remained unclaimed and France preferred to invoke Article 1382 of the French Civil Code[94] before the US Courts instead of accepting the remedies under the Civil Liability Convention. Since the principal office of Amoco was in Chicago, the cases had to be brought before the US District Court of Illinois.[95]

A first lawsuit was filed by the French government to claim compensation for clean-up costs and pollution damage. A second group of lawsuits was brought by local municipalities (*communes)* associated with individuals and associations (hotels' owners, fishermen and oystermen) who lost their businesses due to the oil spill. They all together asked compensation for environmental and private damage. To be more precise, environmental damage represented only one part of the total amount of nonmarket damage caused by the accident. The other part was represented by the reputational damage of coastal towns, villages and commercial activities (*perte d'image de marque*).

[91] Laubier, above n.86, at 282.

[92] NOAA (National Oceanic and Atmospheric Administration), *Assessing the social costs of oil spills: the Amoco Cadiz case study*, US Department of Commerce (1983).

[93] More precisely, the French government sent Amoco Cadiz an official letter of 'mise en demeure' asking to undertake clean-up operations and to remove the oil. The insurance company responded on behalf of Amoco that the tanker owner was unable to provide personnel and equipment for an accident of such scale. It therefore asked the French State to begin the clean-up and that 'reasonable costs' agreed upon between the French government and the tanker owner would be then repaired. Such agreement was never adopted. See *Matter of Oil Spill by the Amoco Cadiz*, 954 F.2d 1279, 1310 (7th Cir. 1992).

[94] "Every act of a man which causes injury to another obliges the one by whose fault it occurred to give redress.".

[95] United States District Court for the Northern District of Illinois Eastern Division, 1988, in re oil spill by the "Amoco Cadiz" off the coast of France on March 16, 1978.

Set aside the moral damage, the case was particularly interesting as to the valuation of the environmental damage, because of its high magnitude.[96] While the Amoco's responsibility was not so much in discussion,[97] the core of the debate was around the costs to be considered for indemnification and the methodology to assess them. Two main methods came in the spotlight. The first one was to quantify environmental damage based on the sum of current and future expenses to restore coastal habitats and to rehabilitate damaged species.

The second method was to quantify the lost biomass or the total volume of organisms lost in the affected area (based on a sample from the communities in the areas not affected by pollution) and to convert then the resulting volume (260,000 tons) in monetary terms by means of the average market prices of the closest commercialized relatives used as a reference.[98] For instance, the value of all lost shrimps was calculated by averaging the market prices of all species of shrimps that are lost. Apparently, this method of the lost biomass was used by US Courts already before the Amoco Cadiz[99] and again by French Courts in the 1980s after the accident.[100] The French State proposed to use this method, leading to a final claim of about 1.5 billion French francs for environmental damage. Bonnieux and Rainelli, who provided independent expertise on the case, heavily relied on the method of the lost biomass proposed by the marine biologists on behalf of the claimants. The two economists argued that the approach had no economic sense since the price was not the result of a market mechanism (the interplay between offer and demand), thus fully arbitrary.[101] The experts on behalf of the French State counterargued that valuing noncommercial species which were void of recreational value was also outside the economic sphere and, in 1988, the US Court ultimately

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96 The accident caused a considerable loss of marine intertidal biomass that resulted in ecological imbalance. See L. Laubier, 'Ecological impacts', in V. K. Tippie and D.R. Kester (eds*), Impact of marine pollution on society*, at 93ss (1982).

97 The Courts clearly stated that it was based on negligence. It resulted from investigations that oil leaks on the hydraulic oil pipe have been noticed for some months.

98 This study was developed by Claude Chassé, researcher at the Université de Bretagne Occidental (institute d'etudes Marines). It was the first study used by the French party in litigation to claim ecological damage. See C. Chassé, 'The ecological impact on and near shores by the Amoco Cadiz oil spill', 9(11) Marine Pollution Bulletin 298 (1978). It should be noted that counting dead bodies was not possible anymore after several months from the accident. For this reason, Chassé thought it would be better to count the number of subjects in close communities in representative samples and then extrapolate them to the total surface area. This method relies on two basic assumptions: the effect of the toxic oil is the same in the total area and the composition/abundance of flora and fauna is constant in the whole area. Solicitors from the opposite side clearly argued against these assumptions as main methodological weaknesses of the lost biomass method.

99 See E.H.P. Brans, *Liability for Damage to Public Natural Resources: Standing, Damage and Damage Assessment*, International Environment Law and Policy Series, Vol. 61, The Hague, Kluwer Law International, 2001.

100 For the references to the cases, see A. de Raulin, 'L'Épopée Judiciaire de l'Amoco Cadiz', 120 *Journal de Droit International* 41, at 78 (1993).

101 F. Bonnieux and P. Rainelli, 'Évaluation des dommages des Marées Noires: Une Illustration à Partir du Cas de l'ERIKA et des Pertes d'Agrément des Résidents', 357 *Economie et Statistique* 173, at 174 (2002). Laubier (above n. 86, at. 285) adds that the Court based its decision on a previous case in the US (the shipwreck of the Zoe Colocotroni in Puerto Rico in 1973) that achieved an extremely high monetary claim thanks to the biomass method.

rejected the biomass method because speculative as well as related to res nullius.[102] The Court explicitly refused compensation for biomass that no-one valued but it agreed on making people responsible for the cost of restocking the species affected by pollution. Consequently, halfway the discovery process, the French plaintiff abandoned the biomass method and replaced it with one based on the costs of programmes of future ecological repopulation that brought to claim 587 million French Francs. The defendant attacked both the expertise of the scientists calculating these costs and the fact that some of the costs were already covered by a regional programme of development of Aquaculture in Brittany. Drawing on these arguments, the US District Court of Illinois rejected also the new claim for two main reasons: the marine environment already recovered from natural degradation during the previous ten years (ten years already passed since the accident) and, secondly, the plan of restoration put forward by France seemed to be aimed at improving the environment for the commercial interests of the local fishermen rather than simply returning it to the baseline (strategic use of the method).[103] Eventually, the US judges sentenced the polluter to refund the French government for just a part of the total clean-up costs , limited to what could be proved by invoices (the only documents that the claimants could provide in litigation). [104] The final award in 1988 was 61 million French Francs (90 million with interests) that was increased to 123 million French Francs with a 1991 decision[105] and to 226 million French Francs in 1992 after appeal.[106] Some commentators stated that this change in the end was clearly influenced by the settlement of the Exxon Valdez in October 1991 and the evident disproportion between the two monetary awards for environmental damage.[107] The next section will shed a light on this point.

Lastly, on the compensation of environmental damage, it must be mentioned that the Ligue de protection des oiseaux (League for the Protection of Birds) also claimed compensation for the costs to re-establish the lost population of puffins after the oil spill. The US Court rejected the claim for compensation for a ‘repopulation programme’ (i.e., introduction of new species) arguing that the program was not yet started and that its goal was effectively to improve rather than repairing the environment. US judges accepted the claim for restoration costs raised by

[102] Oil Spill by the Amoco Cadiz, 1988, at 23. The Court stated that the reparation of ecological damage was not possible under the law because no property rights could be identified upon the damaged ecosystems (res nullius). Also, the Court questioned the validity of the biomass method (like in the previous case of the Zoe Colocotroni) because complex, speculative and based on a chain of assumptions where the deficiency in any one would dramatically affect the final results.
[103] Fourcade, above n. 85, at 1755.
[104] The Court rejected compensation for all clean-up costs that were not well documented and/or that seemed to be affected by mistakes in calculation. “A victim of a tort may not make a profit on the transaction”. See *Matter of Oil Spill by the Amoco Cadiz*, 954 F.2d 1279, 1310 (7th Cir. 1992), par. 99ss.
[105] The French State resorted to a correction procedure to bring factual elements and to ask for mistakes to be corrected.
[106] The Court of Appeal corrected the calculation rate of the interests from 7.2% per year to 11.9% per year.
[107] Laubier, above n. 86, at. 286.

the environmental association limited to the already incurred rehabilitation costs.[108] From an economic standpoint, this argument put forward by the Court may not lead to efficiency because it is quite hard to mark a distinction between reparation and improvement, especially in open environments.[109] Additionally, the dynamism of ecosystems makes more difficult to establish whether the environment had been fully repaired. So the requirement of reparation might lead to extremely narrow decisions.

The debated Issues brought Bonnieux and Rainelli to state that: 'In a disaster such as the Amoco Cadiz oil spill, one of the most surprising facts is the important gap which exists between efforts made by the economists to quantify the losses and the damage awards calculated by the Court'.[110]

In conclusion, the compensated environmental damage consisted of part of the total estimated clean-up costs and restoration costs only limited to damage reparation. Yet, it was what Fourcade called 'a protracted and hugely expensive international legal battle'[111] where the high litigation costs not only matter for the general efficiency of the legal system but also for the optimal internalisation of environmental damage in the assessment. On this point, Fourcade reported that 'the legal and expertise costs incurred during the trial were so enormous that the syndicate of communes found itself several times on the brink of bankruptcy. In one particularly challenging episode, the French government bailed out the penniless syndicate; in exchange, the syndicate agreed to abandon the ecological damage claim during the appeal'.[112]

Therefore, if the scientific demonstration and valuation of environmental damage requires a very high level of financial commitment that is often unattainable by private parties and small collective groups (like the municipalities and environmental associations in the Amoco Cadiz case), these actors would be often cut out the legal arena even if they have legal standing. This was a crucial point in the Amoco Cadiz further accrued by the fact that the trial was taking place in the US and not in Europe. Lacking adequate evidence to challenge the Amoco Corporation and given the 'adversarial nature' of the discovery process in common law countries, the outcome of the environmental damage assessment was obviously more

[108] That was a very small amount of money, 'peanuts' in the words of a person interviewed by Fourcade. See Fourcade, above n. 85, at 1755, footnote 26.
[109] Bonnieux and Rainelli (2005), above n. 17.
[110] Boinneux and Rainelli (1993), above n. 44.
[111] Fourcade, above n. 85, at 1745. The Court in the end stated: 'The trial on damages lasted longer than the trial on the merits. More than a year of trial time was spread over about three years--some before Judge McGarr left the bench, some after. Two principal damages opinions span 575 pages, and there were many supplemental opinions and orders. About a dozen issues remain in dispute.' (*Matter of Oil Spill by the Amoco Cadiz*, 954 F.2d 1279 (7th Cir. 1992), section VII).
[112] Fourcade, above n. 85, at 1758, citing at footnote 32 the Amoco Cadiz final decision of 1992 where the Court explicitly admitted that the ecological damage claim might have been compensable.

favourable to the polluter and less favourable to the victims of environmental damage. Those who were legally entitled to claim compensation for environmental damage (including non-use values) in France and with the highest interest to quantify them in an accurate way (i.e., not limited to clean-up and restoration costs) were also the least able to afford the costs involved in such environmental damage assessment. Therefore, the Amoco Cadiz shows also the crucial role of technical and financial barriers to entry in litigation for the optimal internalisation of environmental damage in liability lawsuits. The Exxon Valdez, in the following paragraph, shows instead how the existence of a public trustee (combined with other socio-cultural factors) can help overcome such barriers.

8. The Exxon Valdez case (1989)

On 24 March 1989, the American 1986-built super-tanker Exxon Valdez went aground on Bligh Reef near Valdez in Alaska after leaving its loading point and hitting a rock. 42,000 tons of light crude oil (12 million gallons), comparable to the oil transported by the Amoco Cadiz for toxicity and chemical properties, spread out rapidly on the surface of the vast Gulf of Prince William.[113] The oil reached and affected 1,500 miles (more than 2,000 km) of cliffs and beaches that had to be closed to fishing, boating and surfing for one year. The environmental damage amounted to 350,000 seabirds dead, plus 2,800 sea otters, 300 harbor seals, 10 sea-lions, 250 bald eagles, up to 22 orcas and billions of salmons and herring eggs, whose reproduction time in the year lies between March and April.[114] On top of everything, the accident damaged an area of peculiar natural beauty, ecological diversity and symbolic value for Americans that further accrued the dimension and public outrage for the event. The Exxon Valdez represented the largest oil spill in US waters until the Deepwater Horizon accident happened in 2010 (see chapter III).

8.1 The clean-up

The authorities decided to avoid chemical dispersants and to employ techniques of recovery at sea with nets (given that the sea was usually calm and sheltered by the islands) and techniques

[113] G. Shigenaka, 'Twenty-Five Years After the Exxon Valdez Oil Spill: NOAA's Scientific Support, Monitoring, and Research', NOAA Office of Response and Restoration (2014), at 1.
[114] Numbers of dead animals (best estimates) represent the measure of biological impact of an oil spill. Reported data can be found in Shigenaka, above n. 113, at 7.

of pressure-washing with cold or hot water ashore, complemented by bioremediation on beaches. The latter aimed at stimulating the growth of local bacteria that could damage the hydrocarbon molecules. Apparently, natural cleaning of the subsurface sediments occurred within three years 'on the finer-grained gravel beaches that have steeper slopes, a thin sediment veneer over the rock platform, and no surface armoring'.[115] The total cost of clean-up was refunded by the insurers and the Exxon corporation (more than $2 billion in total) and that happened very quickly through the out-of-Court settlement in 1991. Moreover, the Exxon Corporation immediately after the accident hired and paid 11,000 people to clean the coast. Arguably, clean-up costs in the Exxon Valdez were 100 times the costs of clean-up in the Amoco Cadiz oil spill which spilled more than six times the amount of oil in Prince Sound Alaska.[116] Moreover, the French government in the Amoco Cadiz was refunded only fourteen years after the accident with the final judicial decision in 1992.

Despite the intensive clean-up effort, it must be pointed out that not everything went back to the conditions prior to the accident. Fourteen years after the accidents it was reported that the number of two species greatly affected by the oil spill (the sea otter and the harlequin duck) considerably decreased, hence showing an unexpected linkage between the ecotoxicological aggression and populations' dynamics.[117] A possible reason for that is the fact that the molluscs on which these species feed themselves were loaded with the hydrocarbons trapped in deep sea sediments where those molluscs live buried. Toxic food might have had an impact on the fertility of the two species, further worsened by the slower hydrocarbon's biodegradation in colder waters. Likewise, the number of herrings born from eggs hatched during the spill in 1989 was considerably low after four years in 1993 (herrings of that specific species are exploited from the age of 4).[118]

8.2 The litigation and the natural resource damage assessment

Immediately after the accident, the Exxon Corporation pled guilty and decided to bear the whole responsibility for the accident. Plausibly, either the known legal saga of the Amoco Cadiz or the compact league of claimants (Alaska authority, NGOs, Alaskan citizens and Indian

[115] O.M. Hayes and J. Michel, 'Factors determining the long-term persistence of Exxon Valdez oil in gravel beaches', 38(2) *Marine Pollution Bulletin* 92 (1999).

[116] Fourcade calculated about $50,000/ton of oil in the Exxon Valdez against $545 in the Amoco Cadiz (see M. Fourcade, above n. 85, at 1744).

[117] Laubier, above n.86, at 288.

[118] For more details, see Laubier, above n. 86, at 289.

communities) played a role for taking such decision.[119] Moreover, the State of Alaska managed to receive from the Federal Government $35 million to investigate and litigate natural resource damage. In the end, it spent up to $67 million on researching and documenting the various types of economic, social and environmental damage caused by the spill.[120] The resulting amount of scientific evidence on the economic value of the damaged natural resources was crucial for the following negotiations with the Exxon Corporation and especially the $1.2 billion compensation agreed only for natural resource damage. As provided by US laws, the trustees[121] invited the Exxon Corporation to conduct the natural resource damage assessment in cooperation (joint damage quantification and agreement on optimal primary and compensatory restoration options[122]). This cooperative procedure aims to minimize both assessment and litigation costs, since it accelerates restoration completion and it facilitates settlement negotiations.[123] Therefore, in 1991, just one year before the Amoco Cadiz final decision, the Federal State, the State of Alaska and Exxon achieved an out-of-Court settlement approved by the US District Court, according to which Exxon had to pay:

- $2 billion to clean up the area ($1.2 paid by Exxon and $1 paid by the insurers);
- $300 million for compensation of private economic damage;
- $900 million (payable over 10 years) for compensation of the damage to public natural resources + $100 million for environmental restoration ($1.025 billion);
- $125 million in criminal sanctions for the environmental crime;
- $500 million in punitive damages.[124]

By summing $2 billion for clean-up and $1billion for environmental liability, the total environmental damage paid by the Exxon Corporation was around $3 billion. The disproportion with the Amoco Cadiz is evident considering that the Amoco spilled 228,000 tons of oil over 300 km of coast and the Exxon spilled 42,000 tons of oil on something more than 2,000 km of coast.

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[119] Laubier, above n. 86, at 287.
[120] Fourcade, above n. 85, at 1759.
[121] In the US, the public trust doctrine specifically applies to public natural resources (air, rivers, oceans, lakes, groundwater, public land). Since the 1960s, numerous nature management agencies have been established to implement nature protection laws and they have been made 'trustees' for the specific natural resources of which they are in charge. One of their missions is to seek compensation for environmental damage. The National Oceanic and Atmospheric Administration (NOAA) is the main trustee at the national level. Many other trustees work at various levels of government and they need to coordinate their actions if they share responsibilities for the same resources.
[122] It might be useful to recall that primary restoration is aimed at restoring the damaged resources, while compensatory restoration at finding restoration options that compensate for interim losses until full restoration and permanent losses.
[123] On this point, see S.M. Thur, 'Resolving Oil Pollution Liability with Restoration-based claims: the United States experience', 32(3/4) *Océanis* 375 (2006).
[124] The final verdict on the case was delivered by the US Supreme Court on 26 June 2008.

Concerning the method of environmental damage assessment, the District Attorney's Office, thanks to the large sums invested in litigation, could commission some environmental economists to assess the non-use damage through contingent valuation,[125] knowing that this method would allow to attain a higher level of damages and so 'make the public whole', which in the US is equivalent to 'compensate the victims'.[126] The results of this study are in Carson et al., 1992.[127] The use of this method was perfectly in line with the state of the art in environmental economics in the US at that time, with the evolution of the law and the caselaw and with the financial and technical capacity of public agencies.[128] By asking people to provide a monetary equivalent of their utility loss and then aggregating the individual preferences, the CV method allowed to reconstruct the missing demand curve for non-commercialised environmental goods. Yet, the validity of the method was still much debated and, for this reason, the State of Alaska hired some of the most reputable economists, such as the Nobel Prize Winner Robert Solow.

The Exxon Corporation did the same, by hiring the Nobel Prize Winner Kenneth Arrow as a scientific consultant. The company also sponsored a symposium on contingent valuation and a book.[129]

The economic team working for the State of Alaska estimated the environmental damage around $2.8 billion ($31 for the WTP of the median American household for a programme that would prevent a similar accident in the future, multiplied by 91 million households), which was much more than the mere costs of replacement.[130] This claimed sum was then settled for $900 million, plus $100 million for unforeseen long-term environmental damage and the sum was justified by the US State before the public by drawing a parallel with the ongoing Amoco Cadiz litigation: 'Although the Exxon Valdez oil spill was one-sixth the size of the world's largest, involving the Amoco Cadiz, Exxon is paying over six times the amount awarded to the French plaintiffs after 12 years of litigation (…). The proposed settlement is thus advantageous

[125] See chapter II for the notion of CV and chapter V for the legal background (Ohio Decision).
[126] Fourcade, above n. 85, at 1759.
[127] R.T. Carson, R.C. Mitchell, W.M. Hanemann, R.J. Kopp, S. Presser & P.A. Rudd, 'A Contingent Valuation Study of Lost Passive Use Values Resulting from the Exxon Valdez Oil Spill', Report to the Attorney General of the State of Alaska, San Diego, California (1992). See also R.T. Carson, R.C. Mitchell, M. Hanemann, *et al.,* 'Contingent Valuation and Lost Passive Use: Damages from the Exxon Valdez Oil Spill', 25 *Environmental and Resource Economics* 257 (2003).
[128] See chapter V on this evolution.
[129] The procedures of the symposium are in 'Contingent Valuation: A Critical Assessment: Proceedings of a Symposium Held in Washington, 2-3 April, 1992, Cambridge Economics (1992). The book is instead: J.A. Hausman, *Contingent Valuation: A Critical Assessment* (1993).
[130] For the exact costs of replacing lost animals, see G. Brown, 'Replacement Costs of Birds and Mammals', Report to the Attorney General of the State of Alaska, cited by Fourcade, above n. 76, at 1762.

not only because of its size, but also because it has been achieved promptly, avoids litigation risks and provide adequate funding for the environment at the time it is needed.'[131]

Admittedly, the final payment of $3 billion for environmental damage and clean-up approximated the $2.8 billion study's estimate by Carson. Then, almost all the damage for natural resources paid by the Exxon corporation went then to finance ecological prevention and monitoring rather than new schools and buildings, as in the Amoco Cadiz.[132]

The Exxon Valdez Oil Spill (EVOS) Council Trustee was formed in 1991 to use the $900 million civil settlement to bring the injured environment back to a 'healthy and productive ecosystem', by implementing a multitude of activities: natural recovery, monitoring, resource and service restoration, habitat acquisition, resource and service enhancement, replacement, meaningful public participation, fiscal accountability. The Council was funded with the invested earnings of the Exxon Valdez Oil Spill Investment Trust fund endowment. Starting from 1993, 18% of the funds were spent on scientific research and a bit less than 40% on the purchase of lands from Native Americans in Alaska for 'habitat protection'. [133] Studies were mainly conducted on an 'ecosystem level' in order to analyse not just the mortality of single species, but the impacts of the spill on entire tropic chains. This shift in the object of ecological studies considerably increased the scientific understanding of post-spill consequences and the attention for species, whose vulnerability was previously underestimated because unknown.[134] It also led the foundations for a deeper understanding of the 2010 oil spill's impacts and more environmental awareness (see later).

Furthermore, the case triggered the revision of the Oil Pollution Act (OPA) in 1990 in order to improve the post-spill management, help reduce oil spills and ensure restoration. For instance, natural resources damage was included in compensation regardless their monetary nature.

To conclude, the Exxon case shows how the use of nature valuation methods that measure non-use values, might result in more scientific knowledge and better management of the injured resources.[135] However, whether this can induce better deterrence of future environmental accidents is a different issue to prove. For sure, the Exxon Valdez shows that cooperation between trustees and polluters can considerably save time and money for both the completion of restoration and the achievement of out-of-Court settlements. Also, it must be pointed out

[131] US District Court of Alaska, Government's Memorandum in Support of Settlement and Consent Decree (1991).

[132] Fourcade, above n. 85, at 1748.

[133] This policy triggered intense conflicts with Natives and they are still ongoing. See, for instance: https://www.chugach.com/tell-the-exxon-valdez-oil-spill-trustee-council-to-stop-the-spend-down/ [accessed 20 November 2023].

[134] Fourcade, above n. 85, at 1766.

[135] This would go against the criticism for which putting a price on natural resources would mean 'profaning nature' (ibid.).

that the procedure of environmental damage assessment in the Exxon case was made possible by specific cultural, technical, financial and institutional factors, whose replicability in other countries can be supported but caution is advised.

The analysis above seems to support at least one main conclusion: adopting a broad or narrow notion of environmental damage (i.e., including or excluding non-use values) should not be treated as a matter of principle but it should be rather subject to strong economic scrutiny.[136] The point is not about monetising the environment or not, but identifying the most cost-effective tools to minimise the costs of accidents, including environmental costs, within certain social, cultural, political and institutional frameworks, considering conflicting private interests, on one side, and the state of the art in ecology, on the other. The next cases support this conclusion, but before continuing with their analysis, the table below provides an easy-to-read summary of the remedies adopted after the two spills considered so far.

	Exxon Valdez 24 March **1989**, Alaska (US) 42,000 tons oil spilled	Amoco Cadiz 16 March **1978**, Brittany (France) 228,000 tons oil spilled
Main legal outcomes	Out-of-Court settlement Alaska-US/Exxon in **October 1991**	Final judgement 7th US Circuit Court of Appeals in Chicago in **January 1992**
Clean-up costs	\$2 billion + (delayed) interests (\$50,000/ton of oil)	(more than) \$124 million (\$545/ton of oil)
Who paid clean-up	Insurers (less than \$1 billion) and Exxon Corporation (\$1.2 billion)	French government and municipalities
Criminal sanctions	\$25 million + \$100 million as criminal restitution for injuries to fish, wildlife and lands	No
Civil damages	\$300 million for private damages (commercial fishermen) \$20 million with Alaska natives	\$61 million + (delayed) interests for clean-up costs to French govt (84%), local municipalities and private claimants (14%)

136 After interviewing ecologists and public officials in Brittany after the Amoco Cadiz spill, Fourcade found a general suspicion against economic methods of valuation or, in her words, a 'visceral anti-economic bias' (Fourcade, above n. 85, at 1767). Also, this bias can be instrumentalised to gain public support for policies and decisions that were instead failures in terms of public money.

Environmental damages	$900 million paid to a trust fund for 10 years + possibility for the State to claim additional $100 million for environmental restoration ($1 billion total)	
Punitive damages	$500 million	No
Additional policy changes	Revision of the Oil Pollution Act (OPA) in 1990 (Public Law 101-280, 101st Congress, 18 August 1990)	Change of navigation routes around Brittany to make them safer Change of international insurance rules to include larger oil spills
Legal framework	American tort law	French tort law + US procedural law (American discovery)

Table 2 [Comparison of remedies of two oil spills][137]

9. The ERIKA case (1999)

The ERIKA oil spill marked the moment when the notion of environmental damage was eventually acknowledged in the French jurisprudence (and later also in legislation) as 'préjudice écologique' and its compensation finally admitted. The accident occurred on 12 December 1999 in Brittany. On 8 December 1999, the old (25-year) tanker ERIKA, carrying around 31,000 tons of heavy oil, left the port of Dunkerque to move to Livourne and it was in the south-eastern part of the coast of Brittany (Bay of Biscay), when the weather became very bad and the ship split in two parts, which sank a few dozen kilometres from each other. As a consequence, 20,000 tons of oil were spilled in the sea, a much smaller spill than the Amoco Cadiz and the Exxon Valdez. Yet, the social, economic and ecological consequences were serious due to the location (close to a natural reserve) and the toxicity of the oil spilled.[138] It was calculated that the diminution of sale of shellfish was between 30 and 70%, with further effects on commercial activities linked to fishing.[139] Huge economic losses also affected the

[137] The data in the table come from the study conducted by Marion Fourcade between 2002 and 2009 and described with full details at 1740 and 1741 of the cited work (Fourcade, above n. 85).
[138] On the chemical properties of the oil, see Laubier, above n. 86, at 290. Yet, Laubier describes that the very poor solubility of the oil in the ERIKA case, combined with the currents, did not allow to reach high concentrations.
[139] Avis sur "Les causes et les conséquences du naufrage du pétrolier ERIKA" adopté par le Conseil économique et social au cours de da séance du 29 mars 2000, at 22.

touristic sector with unpredictable long-term effects on the reputation of the businesses, hence leading the local communities to engage in promoting their image in France and abroad (further expenses).[140] Concerning the environmental damage, it was estimated that the seabirds in the Biscay Bay were by far the most affected: a number between 64,000 and 125,000 died in the immediate aftermath of the accident. Then, almost permanent deposits of oil on beaches and rocks seriously damaged the flora and the fauna. Yet, lack of data on the state of the environment prior to the accident represented a crucial obstacle to assess the exact amount of damage to the French coasts.[141]

9.1 The clean-up

The fight against the oil in the water started on 15 December 1999. The ERIKA vessel was registered under Maltese flag and chartered by the French oil group Total; the tanker was owned by a Maltese company (Tevere Shipping) and classified under the 'Registro Italiano Navale ed Aeronautica' (RINA). At that time, the post-spill response in France was essentially determined by a governmental piece of legislation of 1997 (*Instruction du Premier du 17 décembre 1997 relative à la lutte contre la pollution du milieu marin*) describing in detail which public authority was in charge of each task and how to proceed in terms of measures to undertake, resources to employ and times to respect. The Instruction also delegated competent authorities to establish clean-up methods and storage spaces. Therefore, response operations at sea were led by the French Naval Command in Brest under the French National Contingency Plan (POLMAR). However, old and never revised plans of local administrations with missing or contradictory guidance made the 1997 national law absolutely ineffective.[142] Reportedly, lack of precise information about the landing area, the chemical composition of fuel oil, where to stock toxic materials were among the main factors inhibiting the success of clean-up.[143] The people spontaneously joined the efforts to fight against the oil spill but they could not decide on the appropriate clean-up methods.[144] In addition to all, bad weather and widespread diffusion of oil led to the recovery of less than 3% of the total volume of oil spilled during the

[140] Ibid., at 23.
[141] 'L'absence d'inventaire systématique et permanent de ces zones ne permet pas une évaluation chiffrée des pertes' (ibid., at 26).
[142] Ibid., at 13.
[143] Apparently, interventions plans establishing storage areas and pollution management were adopted late at the local level due also to the late adoption of recommendations by the Ministry of the Environment (Avis above n. , at 16-17). ''Plus l'intervention est précoce et plus les risques ou l'étendue de la pollution ont des chances d'être limités' (ibid., at 17).
[144] Ibid,, at 13.

first clen-up. Cleaning operations started again in June 2000 with better weather and additional 10,000 tons of oil were recovered. Most of the clean-up was completed in November 2001.
In February 2000 the Ministry of the Environment, together with the interdepartmental Council for the Town and Country Planning, set up an expensive programme for the scientific monitoring of the environment (€4.6 for three years), the chemical monitoring of the ecotoxicological consequences on rocky zones, water, organisms, the marine life and flora (€4.6 million for five years) and the strengthening of previous coastal programmes.[145]
Data on the population of birds after some years showed that the number of the most affected species (sea birds, like the common guillemot) did not decrease significantly in the following two years.[146] The chemical quality of the water was monitored until the end of 2003 and the data led to partial bans for shellfish farming and fishing till the beginning of 2001. In general, a return to normality could be seen after three years from the spill.[147]

9.2 The litigation and the environmental damage assessment

When the ERIKA spill occurred, France was already bound by the International Convention on Civil Liability for Oil Pollution Damage (CLC) and the International Oil Pollution Compensation Fund of 1992 (IOPCF),[148] providing for strict liability of the shipowner and requiring mandatory insurance under a certain limitation. Within this framework, the P&I Insurer paid compensation on behalf of the shipowner up to a total of €129.7 million. However, the injured parties (civil parties under French law) tried to overcome the low limits of liability under the CLC (still low at that time) [149] by starting in 2007 a lengthy litigation process against the shipowner (the director of Tevere Shipping, the director of the ship management company, the classification society (RINA) and the charterer (Total SA.)
The Criminal Court of first instance in Paris found all the four defendants criminally liable for the oil spill damage and civilly liable under French law.
In 2010, the Paris Court of Appeal upheld the decision of first instance but considered that Total was entitled to benefit from a special provision of the CLC (channeling provision) for

[145] Laubier, above n. 86, at 292.
[146] Laubier, above n. 86, at 292.
[147] Ibid., at 294.
[148] See chapter IV, §5.
[149] At that time, the compensation available under the 2000 Fund Protocol was 203 million SDR. As a reaction to the ERIKA spill, the EU Commission proposed to set up a European Compensation Fund with a ceiling of 1 billion. The Council decided to pass on the proposal to the International Maritime Organisation (IMO) that adopted in 2003 a Protocol establishing a Supplementary Fund up to the same amount proposed by the Commission.

which no claim could be made against other parties than the shipowner unless it was proved that they intentionally or negligently caused the damage.

In 2012, the French Cour de Cassation confirmed that all four parties were criminally liable for oil pollution damage. Concerning their civil liability, the Court reversed the appeal decision and found that Total's liability could not be channeled to the shipowner because its lack of care was equivalent to 'recklessness' under the CLC.

Another civil liability proceeding was started by the French municipality of Mesquer against Total SA. for liability under the EU Directive 75/442/EEC (the Waste Directive). This lawsuit went up to the EU Court of Justice. In 2008, the ECJ clarified that the seller-charterer of the ship carrying hydrocarbons (i.e., Total) shall be regarded as 'producer' and 'previous holder' of that waste under the meaning of the EU Waste Directive. Therefore, it has to bear the cost of waste disposal to the extent that contributed to the risk of the pollution event by failing to take precautionary measures (i.e., careful choice of the ship).[150] As a consequence, Total incurred the expenses for waste treatment, recovery of oil from the wreck and other costs.

Having said that, the claims for compensation of environmental damage in the ERIKA case were fragmented among various claimants (municipalities, regions, departments, environmental associations) and calculated according to different methods, like in the Amoco Cadiz case.

The French State only sought compensation for the expenses in personnel and material used to clean-up. Indeed, the French National Contingency Plan did not allow the State to raise a general claim for environmental damage.[151]

The environmental associations (Greenpeace and League for the Protection of Birds) asked compensation for the damage to non-traded natural resources, namely birds. They quantified the claimed amount of money (€1 million) by multiplying the number of birds found dead in the post-spill assessment (150,000) by the price used in French Courts for birds caught after hunting (€75). The total price was then increased in consideration of the vulnerability of the species and the fact that they cannot reproduce in captivity.

[150] Judgment of the Court (Grand Chamber) of 24 June 2008, Commune de Mesquer v Total France SA and Total International Ltd., Case C-188/07, European Court Reports 2008 I-04501.

[151] Further money was injected in the economy by the French government in the form of State Aid. See: Direction des Études Économiques et de l'Évaluation Environnementale, Ministère de l'Écologie, de l'Énergie, et du Développement Durable, 'Le Jugement du Procès de l'ERIKA du 16 Janvier 2008 : Responsabilité Pénale du Pollueur et Préjudice Écologique', Numéro 15, Avril 2008, at 25.

The Association Interrégionale Ouest Littoral Solidaire (union of cities, departments and regions) asked the Institut national de la recherche agronomique (INRA)[152] in Rennes to quantify the damage to the natural resources. INRA came out with a total claim of €371.5 million of which:

- two thirds referred to lost use values for missing recreational activities (mainly, sport, fishing and strolling);
- one third referred to lost non-use values (damage suffered by all the inhabitants of the affected coastal regions for the lost common goods regardless their use).[153]

The assessment concerned two following years (2000 and 2001) based on the assumption made by INRA researchers that recreational activities and the provision of ecological services would come back to normality after two years.[154] The table below illustrates in detail the numbers associated by INRA with the loss of use values (*pertes d'usage*), € 234.6 million, and the loss of non-use values (*pertes de non-usage*), € 136.9 million, in order to obtain the total amount of ecological loss (*prejudice écologique*).

10^6 € 2005	2000	2001	Total
Pertes d'usage (pêche à pied)			
Littoral pollué	134,2	57,5	191,7
Nantes	30,0	12,9	42,9
Reste	?	?	?
Total	164,2	70,4	234,6
Pertes de non-usage			
Littoral pollué	11,5	5,7	17,2
Nantes	6,4	3,2	9,6
Reste	73,4	36,7	110,1
Total	91,3	45,6	136,9
Préjudice écologique			
Littoral pollué	145,7	63,2	208,9
Nantes	36,4	16,1	52,5
Reste	73,4	36,7	110,1
Total	255,5	116,0	371,5

Table 3 [The calculation of environmental damage in the ERIKA case[155]

[152] Since January 2020, the new National Research Institute for Agriculture, Food and Environment (INRAE) replaced the previous National Institute for Agricultural Research (INRA) and merged it with the National Research Institute of Science and Technology for the Environment and Agriculture (IRSTEA).

[153] 'Le Jugement du Procès de l'ERIKA du 16 Janvier 2008', above n. 151.

[154] However, considering that negative consequences would plausibly last more than two years, the researchers considered this amount of damages as minimal value. Plausibly, the choice of two was induced by other considerations, such as the interest of the owners of recreational activities to go back to normality. See on this point Fourcade, above n. 85, at 1767.

[155] F. Bonnieux, 'Evaluation Économique Du Préjudice Écologique Causé Par Le Naufrage De l'ERIKA', Rapport confidential, Unité d'Economie et Sociologie Rurales de Rennes, INRA (2006), at 35.

Lost use values were then assessed by means of a replacement cost approach, while lost non-use values were assessed by resorting to contingent valuation studies already available. The table below summarised the main methods used by INRA, on behalf of the plaintiff:

Valeur unitaire	Méthodologie
Pêche à pied • valeur d'une visite	Méthode du coût de déplacement (données originales)
Pêche à pied • diminution de valeur d'une visite	Méthode d'évaluation contingente (données originales)
Activités de remplacement : • valeur d'une visite	Analyse de la littérature
Protection du littoral : • CAP* des résidents	Analyse de la littérature

*CAP : consentement à payer

Table 4 [Methods to estimate the environmental damage][156]

Lastly, the department of Morbihan employed a method of damage valuation based on a programme of shoreline restoration.[157]

Of all these claims, the Paris Court of First instance only accepted (part of) the claims put forward by the local entities and the environmental associations with a special link to natural resources. More specifically:

- the department[158]of Morbihan, which managed a natural reserve, was entitled to get €1 million for the compensation of natural resource damage. This amount was calculated based on the lost tax income linked to the approval of projects of construction (*permis de construire*), renovation or enlargement of buildings of any nature (*Taxe Départementale des Espaces Naturels Sensibles*).[159] Given that the department would have acquired € 2.3 million per year in the considered period (2000 and 2001) but for

[156] Ibid., at 35.

[157] J. Hay, 'Procès ERIKA: La Question du Prèjudice Écologique', 78 *Journal des Accidents et des Catastrophes* (2007).

[158] In the French administrative divisions, departments represent the entities between the administrative 'regions' and the 'communes'. They replaced the previous 'provinces' and there are now ninety-six departments in metropolitan France and five overseas departments.

[159] The *Taxe Départementale des Espaces Naturels Sensibles* (TDENS) has been introduced within the framework of the general policy of protection, management and public use of sensitive natural areas. Originally introduced in 1985, it is now regulated by Article L142-2 of the urban code (Code de l'Urbanisme, Chapitre II: Espaces naturels sensibles des départments). It represents one of the most important financial resources to manage natural areas of public use by French departments since its introduction in 1985. On the origins of the tax, see J.-L. Lenclos, 'La Taxe Départementale des Espaces Naturels Sensibles', 2 *Revue Juridique de l'Environnement* 189 (1997). For a more updated review, see also C. Delivré-Gilg, 'La Taxe Départementale Des Espaces Naturels Sensibles', 2 *Revue Juridique de l'Environnement* 139 (2006).

the accident and that the pollution affected 662 ha out of 3000 ha (total extent of natural areas), the total amount of damage was € 1,015,066.60 ((2,300,000/3,000) multiplied by 662 and then again by 2 years);

- the League for the Protection of Birds, which acted on behalf of protected birds, was entitled to get €300,000 out of €1 million claimed, based on the replacement costs provided by the National Office for Hunting, plus the expenses in rescuing operations, for a total of €680,000.

Clearly, the final award of compensation for environmental damage (€1,3 million) was disappointing because equivalent to the 0.3% of the economic assessment by INRA and to the 0.7% of the total amount of damages accepted by the Court. Therefore, regions, municipalities (*communes*) and the same League for the Protection of Birds appealed the decision also on the issue of the quantification of environmental damage. On 30 March 2010, the French Court of Appeal with its decision expanded the compensation for environmental damage by awarding the Association Interrégionale with 'damages to the natural patrimony' of more than € 200 million.[160]

Unfortunately, the ERIKA case confirmed what Bonnieux and Rainelli previously stated after the Amoco Cadiz case: "one of the most surprising facts is the important gap which exists between efforts made by the economists to quantify the losses and the damage awards calculated by the Court". Indeed, the final award of environmental damages was still much below the compensation for environmental damage (including non-use values) claimed and assessed by INRA on behalf of the claimants.

Moreover, cleaning operations were delayed and lengthy due to uncertain or missing clear instructions on the division of competences among the public entities involved (like in the Amoco Cadiz case).

Another issue to stress is that the French Courts tried to circumvent the limits to compensation under the international conventions governing oil spills by interpreting the provisions of the CLC in such a way that Total SA. Could bear additional costs. Yet, this came with a lengthy and expensive litigation procedure.

In conclusion, the creation of a notion of compensable environmental damage by French Courts does not seem satisfactory from an efficiency perspective.

[160] For the criteria to assess the 'dommage écologique pur' or 'perte d'amenitè', see Chapitre 3 of the decision at 137ss. The Court rejected the contingent valuation method and opted for a fair overall estimate that took into account various parameters, including the 'importance of the pollution' in the specific area of the claimant. Available at: https://actu.dalloz-etudiant.fr/fileadmin/actualites/pdfs/MARS_2014/ERIKA_CA_Paris_30_mars_2010.pdf [accessed 20 November 2023].

The lack of a clear regulatory framework on clean-up determined a slow response and further costs for the environment. The lack of one single public body entitled to claim and receive compensation for pure environmental damage determined an inefficient multiplication of claims with different approaches and higher costs of litigation. The length of the judicial procedure led to reject compensation for the restoration of ecosystems that naturally recovered by the time of the decision and that, conversely, were not recovered yet.

This apparent mismatch between ecological and legal times (the times for legal compensation do not correspond to the times for restoration) may produce overoptimistic biases of polluters and lead to underdeterrence.[161]

10. The Prestige case (2002)

Although the Amoco Cadiz remains the largest maritime oil spill to ever reach shore[162] and the volume of oil spilled from tankers has decreased since the 1970s (see § 2), large-scale oil spills did not end. The sinking of the Prestige tanker in 2002 unfortunately justifies this statement. It is therefore worth extending the previous analysis to this last case to see whether the final remedies (and the chosen valuation methods) allowed to internalize all the environmental costs.

On 13 November 2002, the 1976-built oil tanker *Prestige* was carrying almost 80,000 tons of heavy fuel oil about 50 kilometers off the coast of Galicia (Spain) when it drifted towards the coast and asked access to a safe haven. Spain, France and Portugal refused access to a sheltered port and the tanker had to be towed out into the ocean. On 19 November 2002, despite the attempts by salvors to reduce the stresses on the vessel, it broke in two and 63,000 tonnes of fuel oil were released into the sea, affecting more than 200 km of coastline. The total damage caused to the economy and the local wildlife was estimated around €4 billion.[163] Given that most of the impact was on the Spanish coast, scholars estimated the short-term and long-term economic damage by looking at the Galician fishing, aquaculture and touristic sectors.[164]

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[161] The full internalisation of environmental damage was not possible because the negative effects on the population of dead birds could only be assessed after five years (time needed for the reproduction). 'Enfin, le délais prévus pour prétendre à una réparation écologique des dommages sont trop limitès: pour certaines espèces, plantes ou oiseaux, il faut attendre cinq à six ans (dix ans prèconisent d'autres experts) pour permettre une èvaluation de leur reconstitution. Le principe de précaution devrai pouvoir répondre à cette particularité. En tout état de cause, on ne peut que constater une distorsion entre temps juridique et temps écologique.' (ibid., at 26).

[162] This is stressed by M. Fourcade, above n. 85, at 1742.

[163] https://www.offshore-energy.biz/spain-to-receive-usd-1-8bn-in-damages-for-prestige-oil-spill/ [accessed 20 November 2023].

[164] Galicia is a Spanish region with an important coastal fishing activity. See: M.D. Garza-Gil, A. Prada-Blanco, M.X. Vázquez-Rodríguez, 'Estimating the Short-Term Economic Damages from the Prestige Oil Spill in the Galician Fisheries and Tourism',

It has been reported that the International Oil Pollution Compensation Fund (IOPCF) office in Spain received claims for a total of €1.037 million and considered only €30.1 million admissible for compensation.[165] The CLC covered up to € 22.8 million and the Compensation Fund up to a total of € 171.5 million.[166] Moreover, since Articles III and V of the 1992 CLC allowed to overrule those limitations of liability in case of damage resulting from personal acts or omissions of other people involved in the accident, the Spanish State tried to sue other parties to obtain a higher compensation.[167]

First, a lawsuit was filed by the Spanish State in the US against the certification company (American Bureau of Shipping). The US Court ruled that no sufficient evidence was adduced to prove that the defendant's breach of duty constituted 'a proximate cause of the wreck of the Prestige'.[168] Moreover, a criminal investigation was initiated in Spain against the captain, the chief engineer and the general director of the merchant navy.[169] The case was referred before the Audiencia Provincial de A Coruña (a northwestern town in Spain), that on 16 November 2013 ruled that none of the parties were criminally liable for environmental damage (only the captain was found guilty of disobedience and condemned to 9 months of prison). Lacking an environmental crime, the polluter could not be civilly liable and the insurer of the navy was also released from any liability. In response to that decision, several agents including the public prosecutor, regional public entities and the civil society appealed the provincial Court's judgement.

The case was finally settled by On 26 January 2016, the Spanish Supreme Court adopted a decision correcting the judgement of the Provincial Court and sentencing the vessel's captain to two years' imprisonment for environmental crime in accordance with Articles 325 and 327 of the Spanish Criminal Code (catastrophic environmental damage).[170] Furthermore, the Spanish Supreme Court affirmed the civil liability of the vessel's captain, the vessel's owner

58(4) Ecological Economics 842 (2006); M.D. Garza-Gil, J.C. Surís-Regueiro, M.M. Varela-Lafuente, 'Assessment of Economic Damages from the Prestige Oil Spill', 30(5) *Marine Policy* 544 (2006); M.L. Loureiro, J.B. Loomis and M.X. Vázquez, 'Economic Valuation of Environmental Damages due to the Prestige Oil Spill in Spain', 44 *Environmental and Resource Economics* 537 (2009).

[165] G. Caballero & D. Soto-Oñate, 'Environmental Crime and Judicial Rectification of the Prestige Oil Spill: the Polluter Pays', 84 *Marine Policy* 213 (2017), at 216.

[166] See the Note by the Secretariat Objective of document: To inform the 1992 Fund Executive Committee of the latest developments regarding this incident. Available at: IOPC/APR16/3/2 - Incidents involving the IOPC Funds – 1992 Fund: Prestige [accessed 20 November 2023]. The compensation available under the more recent regime of 2000 Protocols could reach 750 million SDR. See M.G. Faure & W. Hui, 'Economic Analysis of Compensation for Oil Pollution Damage', 37(2) *Journal of Maritime Law & Commerce* 179, at 202 (2006).

[167] Caballero and Soto-Oñate, above n. 165.

[168] US Court of Appeals, Second Circuit: Reino de España v. American Bureau of Shipping (No. 10-3518), 2011.

[169] For an analysis of the judicial process, see G. Caballero-Miguez & R. Fernández-González, 'Institutional Analysis, Allocation Of Liabilities And Third-Party Enforcement Via Courts: The Case Of The Prestige Oil Spill', 55 *Marine Policy* 90 (2015).

[170] The Supreme Court changed the legal interpretation of the facts already proven in the provincial Court and found the captain guilty of recklessness.

and the British insurer (London P&I Club) based on the exception set forth by Article V, paragraph 2 of the 1992 CLC. According to that provision, there is no limitation of civil liability of the vessel's owner if it is proved that the damage has been caused by his personal act or omission. More precisely, the Supreme Court declared the vessel's captain civilly liable without limitations, the shipowner subsidiary liable (no limitations either) based on the 1992 CLC and Article 120.4 of the Spanish Criminal Code, the insurer directly liable for the maximum of the insurance policy according to Article 117 of the Spanish Criminal Code[171] and the IOPCF civilly liable within the limitation of the 1992 FC. London Club (the insurance company), that was ordered to pay a $1 billion fine over the oil spill, started arbitration proceedings against Spain and France (separately). Two partial awards were issued by the respective arbitrators in the end of 2023 and the appeals are still ongoing.[172] Caballero and Soto-Oñate summarised the change in liabilities from the Provincial to the Supreme Court in an easy-to-read table that might be helpful for this analysis:

	Sentence of the Provincial Court, 2013	**Sentence of the Spanish Supreme Court, 2016**
Captain of the tanker	Guilty of disobedience **9 months in Prison** **No direct civil liability**	Guilty of Environmental Crime 2 years in Prison Direct Civil Liable without limits
Chief Engineer	Acquitted	Acquitted
General Director of the Merchant Navy	Acquitted	Acquitted
Owner of the vessel	Civil liable under the limits of the 1992 CLC: Strict liability, but only until 22.8 million euros	Subsidiary Civil Liable without limits
Insurer of the Vessel	The same liability of the shipowner	Direct civil liable to the maximum amount in the shipowner's insurance policy ($1000 million)
IOPCF	Civil liable under the limits of the 1992 FC: Strict liability from 22.8 million to 171.5 million euros	Civil liable under the limits of the 1992 FC: Strict liability from 22.8 million to 171.5 million euros

Table 5 [Judicial decisions in the Spanish case of the Prestige oil spill] [173]

[171] Article 117 of the Código Penal states: 'Insurers which have assumed the risk of financial liabilities arising from the use or exploitation of any property, industry, undertaking or activity, in the case where the event constituting the risk insured materialises as a result of a circumstance provided for in this Code, shall incur direct civil liability up to the limit of the compensation laid down by law or by agreement, without prejudice to the right of recovery against the person concerned.'
[172] For sake of completeness, it needs to be mentioned that there was also a procedure before the Court of Justice of the European Union (CJEU). With its judgement of 20 June 2022 (case C-700/20), the ECJ held that arbitration could not prevent the insurer from complying with the Spanish judgements and paying € 855 million as compensation for the damage.
[173] G. Caballero and D. Soto-Oñate, above n. 165, at 217.

The main changes therefore concerned: the criminal liability of the captain, the limitation to the shipowner's civil liability and the limitation of the insurer's liability. What about the methodology to assess environmental liability?

Even if the Supreme Court changed the allocation of liabilities, additional issues made it hard to achieve a final quantification of the required compensation. On this point, the Supreme Court tried to overcome the boundaries of the international system and, namely, those set down in Article I, paragraph 6 of the 1992 CLC for which compensation shall be limited to the 'costs of reasonable measures of reinstatement actually undertaken' (or to be undertaken) or the 'costs of preventive measures and further loss or damage caused by preventive measures'. This definition of compensable damage has been defined too narrow in the literature.[174] Furthermore, it contradicts the Spanish legal system and precisely Article 45, par. 3 of the 1978 Constitution (obligation to repair the environmental damage combined with criminal and administrative sanctions upon those who violate the right to enjoy the environment). For this reason, the Public Prosecutor demanded €1,214 million for environmental damage and the Supreme Court accepted the compensation for pure environmental damage in addition to the economically quantifiable losses (costs of reparation, prevention and loss of profits).[175] The quantification of damage was deferred to the Provincial Court of A Coruña. Eventually, in November 2017, fifteen years after the disaster, the Court issued an enforcement order confirming that the captain and the marine insurer were liable and condemned them to pay €1.57 billion of compensation to Spain.[176] On 20 December 2018, the Supreme Court upheld the ruling.

Apparently, the national decisions were heavily criticized also by the IOPCF that insisted on applying the CLC to the case at hand in order to recognize the limitation of liability of the insurer under the CLC and to deny the compensation of pure environmental damage that is not admissible under the CLC.[177]

[174] For a historical review of the definition of pollution damage compensable under the CLC and the controversial exclusion of pure environmental damage, see M. Mason, 'Civil liability for oil pollution damage: examining the evolving scope for environmental compensation in the international regime', 27(1) *Marine Policy* 1 (2003).

[175] G. Caballero & D. Soto-Oñate, above n. 165, at 217.

[176] In 2009, Loureiro and other economists estimated that the total environmental use and passive-use losses caused by the Prestige oil spill were around € 574 million. This was the first CV study in Europe after an oil spill. See M.L. Loureiro, J.B. Loomis and M.X. Vázquez, 'Economic Valuation Of Environmental Damages Due to The Prestige Oil Spill In Spain', 44(4) *Environmental and Resource Economics* (2009).

[177] IOPCF, IOPC/APR16/3/2: Incidents Involving the IOPC Funds-1992 Fund: Prestige (Note by the Secretariat). IOPC Funds Document Services: 〈http://documentservices.iopcfunds.org/meeting-documents/download/docs/4028/lang/en/]〉 [accessed 20 November 2023].

11. Behind the valuation: a sociohistorical perspective

The total amount of compensation paid to the French State in the Amoco Cadiz paled in comparison with the Exxon Valdez settlements.[178] This is particularly striking considering that the Amoco Cadiz spilled around six times the amount of oil spilled in the Exxon Valdez and that the whole litigation procedure lasted many more years. Therefore, it is interesting to dig into the reasons of such differences. Behind all these processes attributing monetary values to goods that are normally unsold in the marketplace, there is what Fourcade called a 'sociohistorical' justification, which is simply the recognition of the link between the valuation and specific circumstances (politics, time, space and social conditions).[179] While it is uncontroversial that the economic valuation of natural resources have contingent sociological underpinnings and it is closely bound with the law, the politics, the experts participating in the valuation, the environmental knowledge of a specific society at the specific time of the accident and the subsequent litigation phase, Fourcade stressed the importance of three main factors:

- the applicable law;
- property rights;
- the public perception of publicly owned natural resources.

As to the applicable law, the Amoco Cadiz followed US procedural law since the case was tried in Chicago. Yet, the case was judged under French tort law. That resulted in American laws forcing French parties to justify with much evidence the claims for environmental damage despite the state itself seemed to be unsupportive of a claim for ecological damage.[180] However, the main difference on the level of substantive law was about punitive damages that are precluded under French law but they are allowed under US law.

Regarding the distribution of property rights, the legal tradition of the 'public trust' plays a role for tidelands and submerged lands that are damaged during oil spills. Traditionally, indeed, these kinds of natural resources are deemed as publicly owned and the US governments had to tackle very rarely private claims on land (e.g., with Natives); citizens are anyway entitled to challenge governments for failing in their public mission. In France, by contrast, land is more tied up with property rights and when it comes to the sea this is regarded as res communis, belonging to everyone and not privately appropriable. Moreover, after the entry into force of a foundational law for the environmental protection in France, either individuals or associations

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[178] 'Amoco got away cheap' (Chicago Sun Times, 12 January 1988, cited by M. Fourcade, above n. 85, at 1745).

[179] Fourcade, above n. 85, at 1724ss.

[180] Fourcade, above n. 85, at 1748.

became both responsible for the state of the environment and at the same time entitled to bring a claim in litigation on behalf the environment.[181] Despite any predictions, financial reasons might create a common situation of freeriding or, conversely, competition between all those acting on behalf of nature and, namely, the state versus the local entities. This antagonism is visible in the Amoco Cadiz oil spill: 72 municipalities banded together to support the region's legal actions to defend interests of local residents and make sure that the money would be distributed across them rather than be centrally appropriated.[182] In that way, Brittany's municipalities obtained compensation for ecological damage consisting of shoreline/tidelands restoration costs, local residents' amenity losses and the moral damage (sufferance for the loss). Also, two environmental organizations (League for the Protection of Birds and Society for the Study and Protection of Nature in Brittany) managed to receive compensation for their work in rehabiliting birds, while the municipalities received compensation for the other noncommercial wildlife. Conversely, in the Exxon Valdez case the US government could claim compensation for the whole injured wildlife. Competing rights of property over natural resources definitely influenced the valuation of natural resources and determined higher costs of assessment. It remains uncertain whether these additional costs helped obtain a more accurate valuation likely to induce a better internalization of environmental costs.

A last factor causing a different valuation of natural resources is represented by the public perception of nature when it is publicly owned and the role of the government in this regard.[183] Drawing on the presumption that nature is basically a human construction that speaks about ourselves as much as about the things we label with that word,[184] Fourcade built a parallel between the emphasis on wilderness in the US and two 'political myths': liberty and the need

[181] 'La protection des espaces naturels et des paysages, la préservation des espèces animales et végétales, le maintien des équilibres biologiques auxquels ils participent et la protection des ressources naturelles contre toutes les causes de dégradation qui les menacent sont d'intérêt général. Il est du devoir de chacun de veiller à la sauvegarde du patrimoine naturel dans lequel il vit. Les activités publiques ou privées d'aménagement, d'équipement et de production doivent se conformer aux mêmes exigences. La réalisation de ces objectifs doit également assurer l'équilibre harmonieux de la population résidant dans les milieux urbains et ruraux.' (Art. 1, Loi n° 76-629 du 10 juillet 1976 relative à la protection de la nature).

[182] It has been reported the existence of strikes and public demonstration against the government, hostility that was plausibly rooted into political divisions between local entities and the right-wing government and further accrued by the revival of the Breton identity. The winning of the national elections by the French socialist party effectively reversed this hostility and laid the foundations for more cooperation between the French state and the local entities. See Fourcade, above n. 85, at 1751.

[183] This is an interesting observation raised by the sociologist Simmel and cited by Fourcade (ibid., at 1728). In short, the exchange process changes the relationship between subjective and objectified values and turns it into something more personal: economic values tend to vest objects with much more value as if they had inherent qualities. In the end, people feel more enjoyment from knowing that something that they own has a high price. The cases considered show that putting a price on the environment has the same effect of 'magnifiying our ecological sensitivity'.

[184] See on this point the groundbreaking work by the environmental historian and former President of the American Society of Environmental History, William Cronon, editor of the book *Uncommon Ground, Rethinking the Human Place in Nature* (1995), and especially his introduction to the book 'Introduction: In Search of Nature' at 23-66, also cited by M. Fourcade, above n. 85, at 1735.

of preserving the frontier in order to keep the 'New World' new.[185] National parks policies set up by US federal agencies further accrued the conceptualization of wilderness as inalienable public good with Alaska as the best example of that. After that the safest and most pristine area in the US experienced an unprecedented oil rush in the 1970s due to the presence of a huge natural gas reserve, the State decided to protect an amount of land larger than California from development, hence making it absolutely priceless, meaning that it could not be sold or leased. Conversely, the French concept of nature is extremely far from the American untamed, publicly owned and priceless wilderness.[186] As a consequence of centuries of small landowners, the industrial capitalism in France can be confronted with the idea of rural civilization rather than pure wilderness. The use of the space to show off military and political power by the absolute monarchy further accrued the notion of nature as human construction. Therefore, in France nature is much more entrenched with rural life and 'man-made' nature rather than the US virgin concept of wilderness.[187] It is possible that these cultural constructions of the natural world somehow influenced the valuation approaches in the post-spill phases of the two accidents. Surely, putting a dollar value on the non-use value of nature is accepted and considered as fair in America as much as it may be looked at with suspicion in Europe.[188]

12. Reflection on the cases: do polluters pay for the full environmental damage?

This chapter focused on four major environmental accidents to see to what extent polluters in real cases were exposed to the full social costs of oil spills.

The starting point was that: 'oil spill prevention, like other non-market goods, is traditionally undersupplied because of the difficulty in observing its value'[189] which only relates to the complex issue of damage valuation. The methods of environmental damage assessment varied from one oil spill to the other. Arguably, the most frequently used method seems to be the 'replacement cost' approach (e.g., number of organisms killed multiplied by the average price of each organism), employed in the Amoco Cadiz and Prestige cases.[190] Another method often used is the 'restoration cost' approach or the cost of measures aimed at returning (or

[185] Fourcade, above n. 85, at 1736.
[186] Fourcade, above n. 85, at 1737-1739.
[187] Ibid., at 1739. Fourcade also noticed that French green theorists do not focus so much on grounding environmental values. They rather tend to focus on how conceptions of nature and human society are intertwisted.
[188] Further empirical research would be needed on this anthropological view but this is outside the scope of this research.
[189] Carson & Walsh, above n. 30, at. 371.
[190] J. Hay and O. Thébaud, above n. 28, at 309. The authors found such a high frequency of employment of the replacement cost approach by comparing the data related to twelve accidents including those selected for this chapter.

accelerating the return of) the environment to baseline conditions or the conditions prior to the accident (in the Amoco Cadiz and the ERIKA cases). The contingent valuation method was only used after the Exxon Valdez spill, because it was permitted by the law, backed by sufficient economic scholarship and the parties had sufficient financial capacity to employ it. [191]

However, five more aspects emerged from the analysis of the cases that may trigger or inhibit the attainment of optimal deterrence, but also cost-effective restoration.

12.1 Clean-up

The first aspect concerns the environmental costs of clean-up. Indeed, both a late response and intensive cleaning may increase environmental costs by delaying full natural recovery or causing long-term ecological damage.[192] Given that different clean-up techniques may cause different impacts on different ecosystems, those taking decisions on clean-up should look not only at the benefits in terms of time and use values, but also at the consequences on the ecosystems, especially on vulnerable ecosystems that may be damaged irreversibly and for which predictions are possible because more data are available. Intensive clean-up is advisable only if the persistence of oil on the shoreline poses a worser threat to human and ecological health than the adverse effects of clean-up itself. Unfortunately, the practice showed that decisions on clean-up may be driven by the private interests of insurers (§4.3), of polluters (§8.1), of public administrations and local users, hence leading to disproportional (Exxon Valdez), inefficient (Amoco Cadiz, ERIKA, Prestige) or late (Exxon Valdez) amounts of money spent on clean-up. In order to make decisions on clean-up more efficient, both the regulatory framework on emergency response actions and the regimes of liability should be improved.

Regulations should introduce properly coordinated response strategies already before the occurrence of oil spills and in a way that the specific conditions of the damaged ecosystems

[191] For other methods employed to assess environmental damage, see A.W. Ando & M. Khanna, 'Natural Resource Damage Assessment Methods: Lessons in Simplicity from State Trustees', 22 *Contemporary Economic Policy* 504 (2004). For instance, in the Seki case (1994, harbour of Fujairah, United Arab Emirates, 16,000 tons of oil spilled) environmental damages were calculated by means of a specific compensation schedule for oil spills that took into account the amount of oil spilled, the type of natural resources affected, their vulnerability and oil recovery actions. Based on this amount, the government of Fujairah submitted a claim to the ship-owner and the insurance company, and the parties eventually reached an agreement. Another method used in the practice (Sea Empress, 1996, harbour of Milford Haven, Wales) has been the 'benefits transfer' that applies unit values from other similar cases. In many cases, estimates have been totally arbitrary, meaning that no justification has been given for the specific amount obtained/claimed. See for more references: Hay and Thébaud, above n. 28, at 307, 310.

[192] It was proved that intensive clean-up after the Exxon Valdez and the Amoco Cadiz significantly delayed the full natural recovery of the environment and heavy clean-up took place also after the Prestige and the ERIKA cases.

are taken into account.[193] Coordinated response plans to oil spills are needed to mitigate the damage and to avoid that the oil spreads over larger areas.[194] Public authorities need to have the mandate to issue orders immediately after oil spills and sufficient personnel.[195] Oil spills in the EU proved that the lack of a response plan giving mandate to a national authority led to an inefficient response that resulted in more environmental damage.[196]

However, regulations may not be enough if competent public authorities do not have adequate incentives to intervene or they do have opposite interests not to intervene (like in the Prestige case). Therefore, liability laws should complement regulations by providing polluters and/or public authorities with optimal incentives to minimise the environmental costs of clean-up.

Nature conservation laws may also work to complement regulations and liability laws for publicly owned natural resources with peculiar characteristics, like protected areas (§4.3).

12.2 Pure environmental damage

The second factor that matters for the full internalization of the environmental costs of accidents is whether 'pure environmental damage' (beyond clean-up and restoration) can be claimed and compensated under liability laws and whether this effectively happens in practice. The four large marine oil spills above showed that claims of compensation for lost non-use values of nature represented the less frequently assessed and compensated share of damages. The pure environmental damage assessed in the Amoco Cadiz case corresponded to the claimed share, but less than the 2% of the estimated amount was eventually awarded. Unexpectedly, neither in the ERIKA accident nor in the Prestige one, damages strictly related to losses of non-use values of the environment were claimed and/or awarded despite they surely occurred. Conversely, in the Exxon Valdez case a high monetary award for environmental (and non-use value) losses was settled in Court. Hay and Thébaud (§6) identified some possible factors playing a role in this regard. First of all, whether liability laws include the non-use values among the compensable heads of damage. Secondly, whether a public legal entity exists that

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193 'Plans should be specific to each area that the tanker travels through and accounts for the different costs associated with the ecosystems in each area'(Carson & Walsh, above n. 30, at. 371)

194 The fact that ship owners spent more money in trying to contain the spill in the very first hours avoided liability for major oil spills.

195 The introduction of prevention plans both locally and nationally in the US that resembled the one described in the contingent valuation study of the Exxon Valdez are correlated with the higher reduction in the number of 'major' oil spills off of North America since the Exxon Valdez. See C. Chapple, 'The 1990 Oil Pollution Act: Consequences for the Environment', paper presented at the Association of Environmental and Resource Economists Summer Workshop, La Jolla, CA (2000). Citation in Carson and Walsh, above n. 30, at. 358.

196 The Prestige tank was towed from port to port while local authorities did not agree on beaching the tank under their jurisdiction.

is legally entitled to claim compensation for pure environmental damage in litigation. However, standing and procedural rights may not be enough. Another crucial factor is represented by the incentive to file a lawsuit and, precisely, whether the expected benefits of claiming compensation for pure environmental damage outweighs the expected costs, taking account of the probability of success in litigation, the length of the procedure, the solvability of the liable party, the technical and financial capacity of claimants.

12.3 Litigation

The third and crucial aspect for the full internalization of the social costs of accidents emerging from the practice is the number and the length of civil liability lawsuits. All the cases above, except for the Exxon Valdez, were followed by civil lawsuits lasting many years until final judicial decision (14 years for the Amoco, 11 years for the ERIKA, 15 years for the Prestige). Although the Exxon Valdez was also followed by several and lengthy lawsuits, the main procedure aimed at compensating natural resource damage and clean-up costs was rapidly settled thanks to the proactivity of the polluter, the limited number of public authorities involved, the cooperative process between trustees and polluter, the incentives to settle, the adversarial nature of the common law civil process and the financial capacity of the parties to hire reputable experts and afford a contingent valuation study that could provide a basis for negotiations. Conversely, on the other side of the Ocean, the length of the lawsuits ended up in the French Court rejecting the claim for restoration costs on the ground that the ecosystems naturally recovered by the time of the decision (see the Amoco Cadiz case). Furthermore, the civil process after the ERIKA spill involved an incredible number of public entities all raising environmental damage claims that in the end were compensated according to equity criteria, a huge wastage of litigation costs. As a consequence, lengthy lawsuits may induce overoptimistic biases of polluters, especially when a restoration cost approach is used to quantify environmental damage. Moreover, they may discourage claimants to file lawsuits given the high expected litigation costs.

12.4 Post-accident activities

The fourth aspect relates to some post-accident factors, such as:

- the efficient distribution of resources between clean-up, restoration and compensation;

- the post-spill monitoring to prevent long-term environmental damage;
- additional remedies to improve prevention of accidents of the same type in the future.

As to the correct allocation of resources to initial response, long-term restoration and compensation, this can trigger optimal damage prevention: it is self-evident that late emergency responses would not be cost-effective. In the Exxon Valdez, the initial under-investment in response was followed by over-investments in later stages. Likewise, a lack of resources invested in long-term restoration may hinder the achievement of full restoration and optimal deterrence.

As to the post-spill monitoring, permanent or irremediable environmental damage was reported after all the considered oil spills (species that could not be recovered anymore: seals and puffins after the Amoco Cadiz, sea otters and harlequin ducks after the Exxon Valdez, permanent deposits of oil on beaches and rocks after the ERIKA) and plausibly further ecological damage remained unnoticed due to the lack of monitoring and/or the current state of the art on the links between ecotoxicology and species population dynamics.[197] From this point of view, the example of the (EVOS) Council Trustee formed after the Exxon Valdez to use the money of the settlement for a range of restoration activities including monitoring was beneficial since it allowed to collect useful data on the state of the ecosystem in the Gulf of Mexico and they turned out useful during the NRDA of the more recent Deepwater Horizon oil spill (see chapter VII). The problem of the choice on how money should be spent could be solved by targeting the research on more complex and vulnerable ecosystems where the damage would be irreversible.

As to the additional remedies, the marine oil spills were often followed by legal changes:[198] the OPA after the Exxon Valdez or the regional fund proposed by the EU after the ERIKA case which led to the Supplementary Fund Protocol at the international level.[199]

12.5 Financial responsibility

[197] On this linkage, see Laubier above n. 86.

[198] See M.G. Faure & W. Hui, 'Economic Analysis of Compensation for Oil Pollution Damage', 37(2) *Journal of Maritime Law & Commerce* 179 (2006).

[199] Indeed, the French Courts in the ERIKA case tried to overcome the limits of the CLC and they were criticized because of this. See J. Hay, O. Thébaud and J.A. Pérez Agúndez, 'Preventing Pollution Through the Compensation for Damage? An Appraisal of the European Experience in the Field of Marine Oil Spills', Post-Print hal-00369490, HAL, 2008. By contrast, in the US, the limits under the OPA can be easily broken (i.e., for a breach of regulation). Also, the OPA does not impede the application of national laws, while this in principle excluded under the CLC in the EU.

Lastly, all the above may not be enough if shipping companies can escape from their liability by declaring, for instance, bankruptcy. Indeed, what typically happens is that large holding companies try to abandon their responsibility every time that the damages exceed the value of the ship.[200] Legal obligations of insurance (or bonds) can effectively reduce the risk of bankruptcy and vertical integration in the organization of oil companies[201] to the extent that the liability of the insurer can cover the damage caused by the spill.[202] International conventions governing oil spills currently tackle this risk as far as financially measurable environmental costs are concerned (clean-up and restoration). Yet, the same safeguard should be needed also for 'pure environmental damage' beyond clean-up and restoration. Additionally, the role of regulations in post-spill response remains crucial to avoid the insurer's moral hazard.[203]

13. Conclusions

While the previous part of this dissertation mainly focused on the methodology of environmental damage assessment, the analysis of four marine accidents with huge ecological impacts in this chapter emphasised additional aspects that equally matter for optimal deterrence and cost-effective restoration. They are represented in the table below:

<table>
<tr><td>CLEAN-UP
Timing
Environmental costs
Private interests</td><td>PURE ENVIRONMENTAL DAMAGE
Legal entitlement to claim compensation
Incentive to sue
Assessment</td></tr>
<tr><td>JUDICIAL PROCEDURE
Length
Number</td><td>POST-ACCIDENT ACTIVITIES
Long-term restoration
Monitoring</td></tr>
<tr><td colspan="2">FINANCIAL RESPONSIBILITY</td></tr>
</table>

[200] The fact that the Exxon Valdez had enough resources to cover the damage was rather exceptional.
[201] This has been found in the US after the 1990 OPA by R. Brooks, 'Liability and Organizational Choice', 45 *The Journal of Law and Economics* 91 (2002).
[202] This means that the insurance contract can limit the liability of the insurer even if the liability of the ship owner is unlimited.
[203] Carson and Walsh, above n. 30, at 368.

The emergence of the five aspects represented in the table supports the theory of optimal remedies proposed in chapter III. Indeed, the (mere) obligation of polluters to restore the environment or the (mere) obligation to pay a certain amount of money may not be sufficient to achieve both the ecological and the economic goal of remedies for environmental harm. The joint goal of efficient internalization of social costs and cost-effective restoration may only be achieved through a combination of remedies providing optimal incentives (also) to clean-up, to claim compensation for pure environmental damage, to adopt judicial decisions within reasonable timeframes and to conduct post-accident monitoring on restoration. Yet, all this may not be enough in the absence of financial guarantees for environmental damage.

Take-aways from chapter VI and bridge to chapter VII

- Current liability laws do not expose polluters to the full cost of environmental accidents, including the pure environmental damage beyond clean-up and restoration costs.
- The long-term environmental impact of clean-up may not be considered by insurers, polluters and public administrations.
- Claims for non-use values of nature are the less frequently assessed and compensated share of damage under any liability regime, except for the US Oil Pollution Act.
- Lengthy lawsuits decrease the likelihood to fully internalise the environmental costs of accidents, whereas settlements for the reimbursement of clean-up costs can make the internalization more efficient.
- Post-spill monitoring is under-supplied but crucial for optimal deterrence and cost-effective restoration.

CHAPTER VII

The Ecosystem Services Approach

This chapter presents a novel approach to non-market valuation proposed by ecological economists and recently applied in a case of environmental damage assessment (Deepwater Horizon). In contrast to traditional economists, who draw on an utilitarian anthropocentric perspective that monetises natural resources in view of their use and transformation, ecologists traditionally adopt a biocentric perspective, which rejects the commodification of the environment and argues in favour of its conservation. Nevertheless, the failure of conservationism in the 1970s, coupled with the mounting demand of the economic system for natural capital, prompted some 'ecological economists' to propose a novel approach that could, in theory, overcome the separation between conservation and development while pursuing 'conservation for development' (Folke 2006). The so-called 'ecosystem services approach' was introduced to emphasise environmental benefits that had traditionally been overlooked in environmental economics. Nevertheless, valuation frameworks of ecosystem services have been systematised according to conventional methods. This chapter therefore sets out to demonstrate how traditional valuation techniques have been applied to ecosystem services and to identify the challenges that have been raised. In particular, it is evident that many uncertainties in the valuation of ecosystem services still occur given the current state of the art in ecology. While a comprehensive and accurate valuation of ecosystem services would be necessary to prevent under-compensation and under-deterrence, the current state of the art on ecosystem services valuation is still quite limited and may be of little assistance in litigation. An exception to this may be represented by wetlands and forests, for which more data and economic values are available in the Ecosystem Service Valuation Database (ESVD). The case of the Deepwater Horizon (BP) is finally reported as a first attempt to apply the ecosystem services approach to damage assessment with limited success.

1. Introduction

According to the definition provided by Robert Costanza et al. in 1997, the term 'ecosystem services' refers to 'the benefits that people derive from functioning ecosystems'.[1] This definition relies upon the word 'ecosystem' that is generally understood to mean a dynamic complex of plant, animal, microorganism communities and non-living environments interacting as a functional unit.[2] Broadly defined, ecosystem services include:

- provisioning services such as food, timber, water and fiber;
- regulating services that affect climate, floods, disease, wastes and water quality;
- cultural services that provide recreational, aesthetic and spiritual benefits;
- supporting services such as soil formation, photosynthesis and nutrient cycling.

It is noteworthy that the concept of nature's service first entered the academic literature in 1977 with the article by Walter Westman: 'How Much are Nature's Services Worth?'.[3] Few years after, in 1981, Ehrlich replaced the original term of 'nature's services' with the current 'ecosystem services'.[4] This new stream of academic papers was the product of growing awareness of the depletion of natural resources in the 1980s. As a consequence of the political debate at that time, a new transdisciplinary field known as ecological economics was created. The aim of the ecological economists was to bridge the gap between ecosystem ecologists and environmental economists. Therefore, the concept of benefits from nature represented the basis for building new scientific literature.

Subsequent to this, twenty years after the early appearance of the concept of nature's services, Gretchen Daily edited the first book on the economic value of ecosystem services.[5] Her aim was to bring together world-renowned scientists from a variety of disciplines in order to assess the condition of ecosystem services in the world and to establish the implications of impaired services for humans. Moreover, in 1997, the first workshop on the total value of ecosystem services and natural capital took place in California.[6]

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[1] R. Costanza, R. d'Arge, R. De Groot, et al., 'The Value of The World's Ecosystem Services and Natural Capital', 387 Nature 253 (1997).

[2] Millennium Ecosystem Assessment (2005).

[3] W.E. Westman, 'How Much Are Nature's Services Worth? Measuring The Social Benefits of Ecosystem Functioning is Both Controversial and Illuminating' 197 *Science* 960 (1977).

[4] For the history of the notion, see H.A. Mooney and P.R. Ehrlich, 'Ecosystem Services: A Fragmentary History', in G.C. Daily (ed), *Nature's Services: Societal Dependence on Natural Ecosystems* (1997).

[5] G. Daily (ed), *Nature's Services. Societal Dependence on Natural Ecosystems* (1997).The book includes case studies to show how different ecosystems provide different services to people.

[6] For a passionate story of these early years, see R. Costanza, R. De Groot, L. Braat et al., 'Twenty Years of Ecosystem Services: How Far Have We Come and How Far Do We Still Need to Go?', 28(A) *Ecosystem Services* 1 (2017).

It must be noted that, broadly speaking the term ecosystem services refers to any biophysical relationships, notwithstanding the typology of impact on people. However, in the field of ecological economics, ecosystem services technically refer only to functions which positively contribute to the human wellbeing. For this reason, some authors argued that one of the main limitations of the ecosystem approach is represented by its inherent anthropocentrism. In the wake of that, recent literature introduced the term 'nature's contribution to people' which includes both beneficial and harmful effects on people's wellbeing.[7]

Despite the existence of alternative views on the values of nature and the human-nature relationship, the purpose of this chapter is to illustrate current methods of valuation of ecosystem services that draw on classical environmental economics. As already discussed in chapter II of this dissertation, all traditional methods present advantages and shortcomings that also apply to the environment viewed as a supplier of ecosystem services. In addition to that, there are challenges and uncertainties specifically related to the ecosystem services approach.

A prior clarification of the underlying reasons of valuing ecosystem services may be needed before delving into the theory and practice of their valuation.

2. Why valuing ecosystem services?

Valuation studies of ecosystem services may be motivated by various reasons. First and foremost, although ecosystems provide a wide variety of services that are essential for humans, only some of them have been priced and incorporated in transactions. Markets are absent for most ecosystem services and this determines the impossibility of including their value in the decision-making. Therefore, the first main reason for valuing ecosystem services is to unveil the impact of human decisions upon the ecosystems and to express these (marginal) value changes in monetary units that can be then incorporated in the decision-making, hence making development policies more accurate.[8] Further reasons for conducting an ecosystem services valuation include the possibility of designing conservation programs based on ecosystem values rather than market prices, the acknowledgement of uncertainty about future demand and supply of natural resources and, finally, the opportunity of including ecosystem values in natural resources accounting.[9]

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[7] U. Pascual, P. Balvanera, S. Díaz, et al. 'Valuing Nature's Contributions to People: The IPBES Approach', 26-27 *Current Opinion in Environmental Sustainability* 7 (2017).

[8] K.T. Turner *et al.*, 'Valuing Nature: Lessons Learned And Future Research Directions', 46 *Ecological Economics* 493 (2003).

[9] U. Pascual and R. Muradian, 'The Economics of Valuing Ecosystem Services and Biodiversity', in P. Kumar, *TEEB, The Economics Of Ecosystems And Biodiversity : Ecological And Economic Foundations* (TEEB), at 192 (2010).

3. Economic valuation of ecosystem services

The underlying assumption behind the environmental economic valuation is that nature is an asset and it holds a value provided that it helps achieve human goals, ranging from aesthetic pleasure to the production of market commodities.[10] As a consequence, the economic value of ecosystem services is a 'marginal' concept, meaning that it is the measurement of changes to social welfare caused by small or 'marginal' changes in the quality or quantity of ecosystems.[11] The economic value is not an intrinsic characteristic of natural resources, but it is assigned by economic agents based on what they would be willing to pay for the services derived from it.[12] The willingness to pay is in turn determined either by the ecological and physical properties of a natural asset or by the socio-economic context of economic agents (human preferences, institutions, etc.).[13] More generally, the economic valuation of ecosystem services is rooted into the preference-based paradigm, which assumes that values arise from individual preferences rather than intrinsic properties of natural resources (biophysical approach to valuation).[14]

Notwithstanding the existence of an important debate between a biophysical and a preference-based theory of value, when talking about economic value we are unanimously referring to the latter axiomatic framework.[15] The following assumption is that ecosystem services are always commensurable in monetary terms and trade-offs in the use of ecosystems can be established by means of money.[16]

Based on these more general distinctions, it is now possible to illustrate with full details how ecological economists framed the economic value of ecosystem services.

4. Ecosystem services under the Total Economic Value (TEV) approach

[10] E.B. Barbier, 'Ecosystems as Natural Assets', 4 *Foundations and Trends in Microeconomics* 611 (2009).

[11] TEEB, above n. 10, chapter 5, par. 8. In the words of Turner, above n. 9: the economic value of ecosystems is the marginal change to economic welfare caused by a marginal change to the state of ecosystems, such as the restoration of a polluted area.

[12] On this, see extensively chapter II.

[13] D.W. Pearce & D. Moran, *The Economic Value of Biodiversity* (1994).

[14] For this reason, the ecosystem services approach builds on the utilitarian perspective like traditional nature valuation methods. See also chapter II, footnote 10.

[15] On the valuation paradigms, see Pascual and Muradian, above n.10, at 193 and E. Gómez-Baggethun and R.S. De Groot, 'Natural Capital and Ecosystem Services: The Ecological Foundation of Human Society', in R.E. Hester and R.M. Harrison (eds), *Ecosystem Services, Issues in Environmental Science and Technology* (2010).

[16] On the use of money as a metric for nature, see also M. Fourcade, 'Cents and Sensibility: Economic Valuation and the Nature of "Nature" ', 116(6) *American Journal of Sociology* 1721, at 1723ss (2011).

The total economic value[17] of ecosystems consists of two main components:[18]

- the 'output value', intended as the aggregated value of the flow of ecosystem service benefits;
- the 'insurance value', which is the capacity of the system to maintain the flow of service benefits notwithstanding variability and disturbance.[19]

The output value can be regarded as the total economic value of ecosystem services and biodiversity. It may be in turn defined as the sum of the values of all services flows (use and non-use values) that are generated by marginal changes of natural capital, now and in the future, and properly discounted. [20] Benefits included in the output value may include the provision of water to households and the industry as well as the mitigation of natural hazards caused by storms.[21]

The insurance value[22] is based on the system's resilience, which is the capacity of the systems to absorb shocks and to self-organize again in order to maintain its essential functions and structure.[23] The ecosystem resilience is therefore what ensures a healthy functioning of ecosystems.[24]

Environmental economists have generally framed ecosystem services as positive externalities that are consumed in the absence of market transactions. Methods to value these 'invisible' benefits from the environment have been developed since the 1960s in order to internalise the externalities in cost-benefit analysis. The assignment of ecosystem services to each component of the TEV has been progressively refined over the last decades mainly thanks to the work of Pearce, Turner and De Groot.

The figure on the next page summarises the types of value of ecosystems within the neoclassical economic paradigm – as it germinated from the work of Krutilla in 1967 (see

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[17] For more information on the total economic value of nature, see chapter II.

[18] Pascual and Muradian, above n.10, at 195-196.

[19] These terms were first adopted by I.-M. Gren, C. Folke, R.K. Turner & I. Bateman, 'Primary And Secondary Values Of Wetland Ecosystems', 4 *Environment and Resource Economics* 55 (1994).

[20] B. Fisher, R.K. Turner & P. Morling, 'Defining And Classifying Ecosystem Services For Decision-Making', 68 *Ecological Economics* 643 (2009). These scholars stressed the need to look at the end products rather than intermediate services in order to avoid double counting of output values.

[21] Pascual and Muradian, above n.10, at 196.

[22] The insurance value is closer to the option value in the traditional TEV framework, see chapter II.

[23] B.H. Walker and J.A. Meyers, 'Thresholds In Ecological And Social-Ecological Systems: A Developing Database', 9(2) *Ecology and Society* 3 (2004).

[24] F. Brand, 'Critical Natural Capital Revisited: Ecological Resilience And Sustainable Development', 68 *Landscape Ecology* 605 (2009).

chapter II) – and related ecosystem services that have been the object of economic valuation methods:

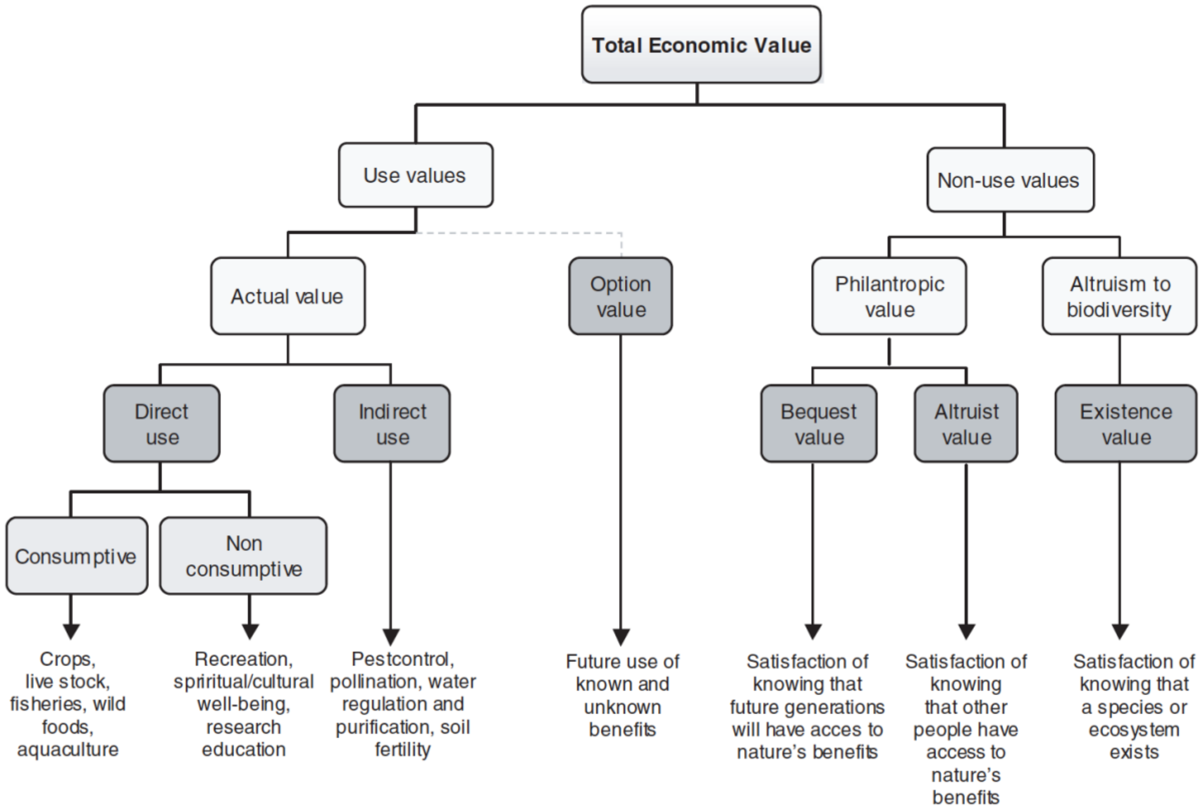

Figure 1 [Value types and ecosystem services within the TEV approach][25]

Use values may refer to the benefits extracted from direct and indirect use of ecosystems. The former (direct use) relates either to extractive (consumptive) use from food or raw materials or non-extractive (non-consumptive) use, such as recreation and aesthetic benefits. The latter (indirect use) generally regards regulating services, like air quality regulation or prevention of erosion. While ecosystem services with direct use share the characteristics of private goods, indirect-use services are generally closer to public services which are not traded in markets.

The option value refers to possibly future ecosystem benefits.[26] Albeit some scholars contested option values,[27] they remain a way to frame uncertainty within the TEV framework.

Lastly, non-use values do not involve any kind of direct and indirect use of ecosystem services. They can be defined as the utility gained by individuals for knowing that ecosystem services

[25] Pascual and Muradian, above n.10, at 197.

[26] J.V. Krutilla and A.C. Fisher, *The Economics of the Natural Environment: Studies in the Valuation of Commodity and Amenity Resources* (1975).

[27] A.M. Freeman, 'The Measurement of Environmental and Resource Values', Resources for the Future (1993).

are conserved (existence value), that other people can take benefit from them (altruist value) or future generations will benefit from them (bequest value).[28]

Given the classification above, it is of great interest to understand the implications of these types of values for the valuation process. In view of that, Table 2 below reverts the approach and clearly illustrates the value categories for each class of ecosystem services:

Group	Service	Direct use	Indirect use	Option value	Non-use value
Provisioning	Includes: food; fibre and fuel; biochemicals; natural medicines, pharmaceuticals; fresh water supply	✓	**NA**	✓	NA
Regulating	Includes: air-quality regulation; climate regulation; water regulation; natural hazard regulation, carbon storage, nutrient recycling, micro-climatic functions etc.	**NA**	✓	✓	**NA**
Cultural	Includes: cultural heritage; recreation and tourism; aesthetic values	✓	**NA**	✓	✓
Habitat	Includes: primary production; nutrient cycling; soil formation	*Habitat services are valued through the other categories of ecosystem services*			

Note: NA = Non applicable.

Table 2 [Valuing ecosystem services through the TEV framework][29]

Based on this table, it is possible to realise how use values relate to provisioning and regulating services for which a market price might be available, while non-use values only refer to cultural services for which a market usually does not exist. Non-use values refer indeed to moral or aesthetic values. In other words, since cultural services are not tangible, their value is more the

[28] This recalls the traditional approach in environmental economics. See C.D. Kolstad, *Environmental Economics* (2000).
[29] Pascual and Muradian, above n.10, at 199.

result of an experience that occurs in the mind of those who value.[30] For this reason, valuing cultural services turns out to be much more challenging than for the other ecosystem services.

5. Valuation methods under the TEV (theory)

Values of ecosystem services under the TEV can be derived from information of individual preferences available in market transactions directly related to ecosystem services or in parallel markets indirectly associated with the service to be valued. If neither direct or indirect price information is available, values need to be elicited through hypothetical markets. In the wake of this reasoning, available methods to value ecosystem services have been categorised into direct market valuation methods, revealed preference methods and stated preference methods.[31]

All these methods have already been examined in depth in chapter II. This section therefore provides a brief overview of their advantages and limitations replicating what has already been said. However, it also emphasises how these methods have been applied on the valuation of ecosystem services in the ecological literature.

5.1 Direct market valuation

Within the TEEB report, direct market valuation approaches to ecosystem services valuation may be based on market prices, costs or production functions.

Market price-based approaches are mainly used for provisioning services, since services hereby produced can be often traded on markets. Market prices are usually considered to provide accurate information on the value of ecosystem services since they embed market preferences and marginal costs of production.[32]

Cost-based valuation[33] is based on the assumption that expenditures on producing and maintaining ecosystem services provide net benefits and these benefits match the original level of benefits.[34] It requires the elaboration of hypothetical scenarios that respond to the question:

[30] Pascual and Muradian, above n.10, at 198.

[31] Y.E. Chee, 'An Ecological Perspective on The Valuation of Ecosystem Services', 120 *Biological Conservation* 459 (2004).

[32] Pascual and Muradian, above n.10, at 199.

[33] The damage costs avoided approach is not a cost-based approach because it is based on the assumption that the cost of damage is a measure of value. See E.B. Barbier, M.C. Acreman & D. Knowler, 'Economic Valuation Of Wetlands: A Guide For Policy Makers And Planners', Ramsar Convention (1997).

[34] Ibid.

what would be the cost to bear if ecosystem benefits had to be artificially recreated? The answer can be searched by looking at:

- the avoided cost or the cost that would have been incurred absent ecosystem services;
- the replacement cost or the cost incurred to replace ecosystem services with artificial substitutes;
- the restoration cost or the cost of restoring lost ecosystem services;
- the mitigation cost or the cost of mitigating the consequences of a loss of ecosystem services;
- the relocation cost or the cost to relocate threatened ecosystems.

Production function-based approaches are based on the assumption that, if ecosystem services physically contribute to the production of other commodities or services traded in markets (e.g., fishing, hunting, farming), changes in ecological functions (improvement or deterioration of environmental quality *e.g.*, water quality) may affect the quantity or price of certain goods and, thus, the consumer surplus.[35] In other words, the flow of ecosystem services ultimately contributes to the economy.[36] For this reason, by modelling the relationship between resources and economic outputs it is possible to elicit the value of non-marketed ecosystem services from marginal changes of economic outputs. According to Barbier, this approach unfolds with two subsequent steps: first the assessment of physical changes in economic activities caused by ecosystem services and, secondly, changes in marketed outputs of traded activities. Finally, it is important to distinguish the marginal value of products from the gross value of output.

5.1.1 Advantages

Direct market valuation approaches are based on data from actual markets. This may have three well-known advantages. First, data on prices, quantities and costs are easy to obtain and less resource-intensive (so, not highly expensive).[37] Secondly, market prices reflect the actual willingness to pay for costs and benefits that are traded, so they are sufficiently accurate. Thirdly, this data are generally regarded as sufficiently objective and thus more reliable than other tools to elicit social preferences. For instance, the cost-based valuation offers a less data-intensive method to estimate the WTP when ecosystem services are not traded and it is easier to measure the costs of producing benefits than the benefits.[38]

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[35] Freeman, above n. 28, at 259.
[36] K.-G. Mäler, I. Gren & C. Folke, 'Multiple Use of Environmental Resources: A Household Production Function Approach to Valuing Natural Capital', in A. Jansson, M. Hammar, C. Folke & R. Costanza (eds), *Investing in Natural Capital* (1994).
[37] G.M. Ellis and A.C. Fisher, 'Valuing Environment as Input', 25 *Journal of Environmental Management* 149 (1987).
[38] Barbier, above n. 34.

5.1.2 Limitations

The main limitation of these approaches is that their applicability relies on the existence of markets where data on prices or costs are available. If markets for ecosystem services or for goods indirectly related do not exist, then the needed data are not available. However, even if markets prices are available, possible distortions may occur due to market imperfections and policy interventions (e.g., subsidies), seasonal variations and other effects on prices.[39] This limitation can be overcome by a data-intensive process of adjusting prices (so-called 'efficiency shadow prices method') in order to match the true economic value or opportunity cost.

Further limitations specifically regard certain approaches. For instance, Barbier warned that the replacement cost method should be used with caution under uncertainty[40] and Daily pointed out that cause-effect linkages between ecosystem services and market commodities are often lacking, so that it is not clear how much of ecosystem services is produced based on a certain change in ecosystem conditions.[41] Lacking direct relationships between resources and economic outputs it becomes more difficult to disentangle effects on production functions determined by interconnected ecosystem services. The risk in the end is of double counting services,[42] especially for multiple use systems.[43]

Cost-based valuation approaches are further limited by the fact that costs are not an accurate measure of benefits.[44] Moreover, restoration costs are limited by the difficulty of restoring previous ecosystem services and replacement costs by the fact that it is difficult not to exceed the original benefits when services are replaced. Likewise, it is unlikely that relocated services can provide the same benefits of lost services in the original location.[45]

5.2 Revealed preference valuation

Revealed preference techniques are based on the observation of preferences shown (revealed) in existing markets which have a correlation with the ecosystem services to value. Two main

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[39] Ibid.
[40] E.B. Barbier, 'Valuing Ecosystem Services As Productive Inputs', 22(49) *Economic Policy* 177 (2007).
[41] Daily, above n. 5.
[42] R. Costanza and C. Folke, 'Valuing Ecosystem Services With Efficiency, Fairness and Sustainability as Goals', in G. Daily (ed), *Nature's Services: Societal Dependence on Natural Ecosystems*, at 49 (1997).
[43] Barbier, above n. 34.
[44] Ibid.
[45] Ibid.

methods are used to elicit revealed preference: the Travel Cost method (TC) and the Hedonic Pricing method (HP).[46]

5.2.1 Advantages

The main advantage of revealed preference consists of the possibility to follow a clear procedure in order to estimate the value of ecosystem services, starting from the search of a surrogate market closer to the environmental goods and services to value, continuing with the choice of the appropriate methodology, the collection of market data, the estimation of the marginal change and ending with the aggregation of values across the population. [47]

5.2.2 Limitations

There are various limitations of revealed preference methods for valuing ecosystem services. First, the application of these methods requires the existence of surrogated markets where values of ecosystem functions are reflected.[48] Secondly, even if it is possible to identify surrogated markets, it is crucial to gather good-quality data and large data sets which may not be always available. Revealed preference approaches are thus data-intensive and time-consuming.[49] Thirdly, market failures, policy failures, income constraints and scarce information may distort prices, hence affecting the accuracy of the elicited values. Fourthly, the validity of the TC method is based on restrictive assumptions about consumer behaviours, such as multifunctional trips.[50]

Lastly, observed behaviours in surrogated markets cannot provide information regarding non-use values which, as already said, represents the biggest share of cultural ecosystem services. As a consequence, these services may inevitably fall out of the estimated values.

5.3 Stated preference valuation

Stated preference approaches are based on surveys that try to elicit preferences on policies that may change the provision of ecosystem services. Three types of techniques fall in this category:

[46] See chapter II for a detailed discussion and more references.
[47] Pascual and Muradian, above n.10, at 201.
[48] A. Kontoleon, U. Pascual & T. Swanson, *Biodiversity Economics* (2007).
[49] Pascual and Muradian, above n.10, at 202.
[50] E.B. Barbier, M.C. Acreman & D. Knowler, 'Economic Valuation Of Wetlands: A Guide For Policy Makers And Planners', Ramsar Convention (1997).

- contingent valuation method, where people are asked – through questionnaires – how much money they would be willing to spend in order to increase the provision of ecosystem service or, alternatively, how much money they would be willing to accept for their loss;
- choice modelling, that tries to model the decision process of individuals in face of two or more alternatives about the services to value;[51]
- group valuation, that combines stated preference techniques with deliberative processes from political sciences in order to capture components of values other than those elicited through surveys.[52]

5.3.1 Advantages

Stated preference methods allows to directly elicit preferences about the values of ecosystem services and to have the best theoretical measures of WTP. Moreover, these are the only techniques to estimate non-use values (option and existence values) and obtain the total economic value.

Furthermore, a CM study allows to estimate marginal values for changes of specific attributes of environmental resources induced by different policies (options). Each option in the survey consists indeed of a different balance of impacts on ecosystems, such that choosing one option rather than another reveals preferences about a specific change of attributes. Also, Adamowicz pointed out how stated preference approaches provide information regarding perceptions, attitudes and previous knowledge.[53] All these additional pieces of information may help better understand preferences for the assessment of ecosystem services. For instance, stated preference may show the relative importance given by respondents to different ecosystem services[54] as well as conflicts among stakeholders and alternative policies options.[55] Lastly, group valuation differs as it has the potential of overcoming limitations of traditional monetary valuation methods.[56]

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[51] The main difference between contingent valuation (CV) and choice modelling (CM) is that in a CV respondents have only one option and they are asked whether they would agree on paying for it or they would rather stick to the *status quo,* whereas in a CM study respondents are given several choices (Kontoleon, Pascual & Swanson, above n. 49).

[52] Spash referred to value pluralism, incommensurability, non-human values and social justice. See C. Spash, 'Deliberative Monetary Valuation And The Evidence For A New Value Theory', 83(3) *Land Economics* 469 (2008).

[53] W.L. Adamowicz, 'What's It Worth? An Examination Of Historical Trends And Future Directions In Environmental Valuation', 48 *The Australian Journal of Agricultural and Resource Economics* 419 (2004).

[54] B. Martín-López, C. Montes & J. Benayas, 'The Role of User's Characteristics On The Ecosystem Services Valuation: The Case of Doñana Natural Protected Area (SW Spain)', 34 *Environmental Conservation* 215 (2007).

[55] P. Nunes, S. Silvestri, M. Pellizzato & B. Voatto, 'Regulation Of The Fishing Activities In The Lagoon Of Venice, Italy: Results From A Socio-Economic Study', 80 *Estuarine, Coastal and Shelf Science* 173 (2008).

[56] R.S. De Groot, M. Stuip, M. Finlayson & N. Davidson, 'Valuing Wetlands: Guidance for Valuing the Benefits Derived from Wetland Ecosystem Services', Ramsar Technical Report No. 3, CBD Technical Series No. 27, Ramsar Convention (2006).

5.3.2 Limitations

Stated preference valuation methods raise concerns in terms of accuracy and validity of estimations which challenge the truth of the estimated willingness to pay. They relate to biases in surveys that limit the reliability of answers,[57] to the discrepancy between willingness to pay and willingness to accept[58] and to the so-called 'embedding bias'.[59] In any case, stating preferences about ecosystem services is as challenging as valuing public goods for which preferences are not well-defined and responses tend to lack sufficient accuracy.[60] Upfront information in questionnaires[61] and valuation workshops held in advance[62] may help respondents to reflect on their preferences and overcome their cognitive constraints during surveys. Likewise, deliberative monetary valuation methods seem to further reduce biases and non-response rates.[63]

6. Valuation methods under the TEV (practice)

While the previous section has provided a theoretical overview of the existing methods to conduct a monetary valuation of ecosystem services, this section shows how the methods have been applied in practice for the valuation of ecosystem services.[64]

The table on the next page summarises which methods are traditionally used to assess the values of nature (not only ecosystem services).

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[57] Barbier, above n. 34.

[58] M. Hanemann, 'Willingness To Pay And Willingness To Accept: How Much Can They Differ?', 81 American Economic Review 635 (1991).

[59] K. Veisten, 'Contingent Valuation Controversies: Philosophic Debates About Economic Theory', 36 *The Journal of Socio-Economics* 204 (2007). 'The embedding effect is the name given to the tendency of willingness-to-pay responses to be highly similar across different surveys, even where theory suggests (and sometimes requires) that the responses be very different', quote from P.A. Diamond and J.A. Hausman, 'On Contingent Valuation Measurement of Nonuse Values', in J.A. Hausman (ed), *Contingent Valuation: A Critical Assessment*, at 1 (1993). The embedding bias occurs when people tend to express the same WTP for an environmental change in a small area as well as in a bigger area because they are truly insensitive to the scope of the survey.

[60] H. Svedsäter, 'Economic Valuation Of The Environment: How Citizens Make Sense Of Contingent Valuation Questions', 79 *Land Economics* 122 (2003).

[61] C. Tisdell and C. Wilson, 'Information, Wildlife Valuation, Conservation: Experiments And Policy', 24 *Contemporary Economic Policy* 144 (2006).

[62] M. Christie, *et al.,* 'Valuing The Diversity of Biodiversity', 58 *Ecological Economics* 304 (2006).

[63] De Groot, above n. 57.

[64] An extensive literature review of studies in this regard can be found in the TEEB report. The authors show how some valuation methods turned out to be more appropriate than others in order to elicit specific value components. This is a consolidated opinion in the literature on the valuation of ecosystem services. On this point, see R.K. Turner, *et al.* 'Economic Valuation of Water Resources In Agriculture From The Sectoral to a Functional Perspective Of Natural Resource Management', Food and Agriculture Organisation of the United Nations (2004).

Approach		Method	Value
Market valuation	Price-based	Market prices	Direct and indirect use
	Cost-based	Avoided cost	Direct and indirect use
		Replacement cost	Direct and indirect use
		Mitigation / Restoration cost	Direct and indirect use
	Production-based	Production function approach	Indirect use
		Factor Income	Indirect use
Revealed preference		Travel cost method	Direct (indirect) use
		Hedonic pricing	Direct and indirect use
Stated preference		Contingent Valuation	Use and non-use
		Choice modelling/ Conjoint Analysis	Use and non-use
		Contingent ranking	Use and non-use
		Deliberative group valuation	Use and non-use

Table 3 [Relationship between valuation methods and value types][65]

By crossing Table 3 with Table 1 (above), it is clear that:

- market valuation has been mainly used to value provisioning services (environmental goods, e.g. fish), regulating services (e.g., flood control) and cultural services (e.g., recreation);
- revealed preference (TCM, HP) to value cultural (e.g., recreation) and regulating services (e.g., clean air, clean water);
- stated preference to elicit social preferences about water quality or recreational activities.

The table below shows the number of studies per category of ecosystem services:

Type of valuation approach	Cultural	Provisioning	Regulating	Supporting
Benefits transfer	9	3	4	6
Cost based	5	27	61	17
Production based	1	33	9	0
Revealed preference	38	18	7	28
Stated preference	46	19	19	50
Grand total	**100%**	**100%**	**100%**	**100%**

Table 4 [Valuation approaches used to value ecosystem services][66]

[65] Pascual and Muradian, above n.10, at 206.

[66] Pascual and Muradian, above n.10, at 208. The numbers in the table refer to 314 peer reviewed valuation studies available at the time of the publication of the report (2010).

The main results from this literature review can be summarised as it follows:[67]

- cultural services have been primarily valued with stated preference methods (contingent valuation for the existence value) and, secondarily, with revealed preference (travel cost method and hedonic pricing for recreational and aesthetic values);
- provisioning services have been mostly valued with the production based or the cost based technique (opportunity costs);
- regulating services primarily with cost based techniques (avoided or replacement cost) and, secondarily, with stated preference;
- supporting services primarily with stated preference (contingent valuation) and, secondarily, with public investments.

The authors of the TEEB did a step forward in the ecosystem valuation by further classifying the previous studies according to the type of biome. Table 5 below shows the distribution of studies on the valuation of forests and wetlands:

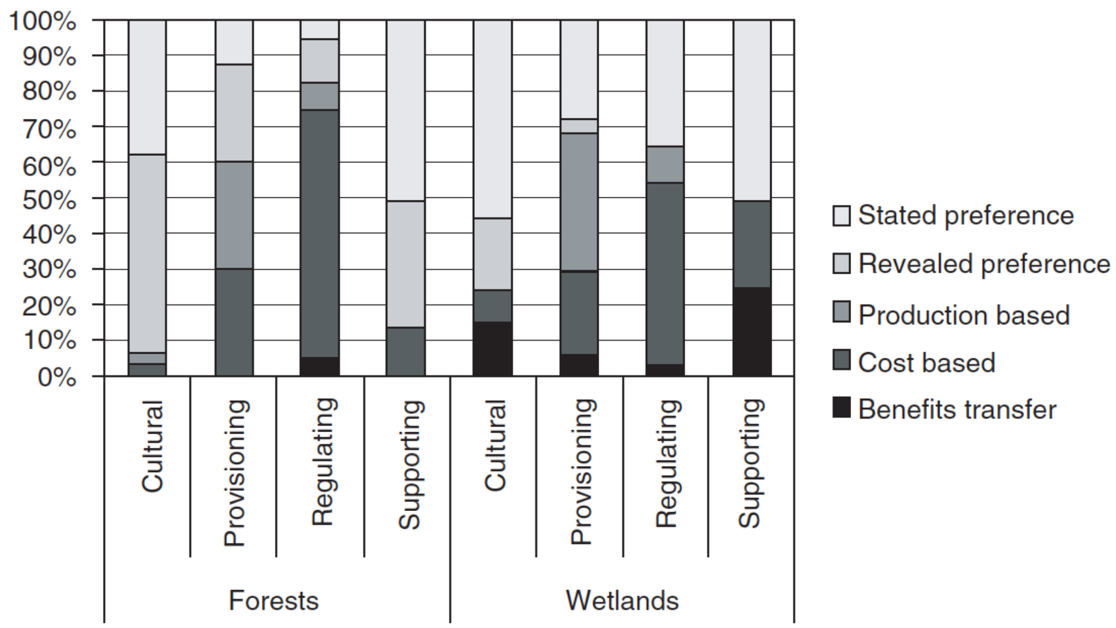

Table 6 [Valuation approaches used to value ecosystem services from forests and wetlands][68]

[67] B. Martín-López, E. Gómez-Baggethun, P. Lomas & C. Montes, 'Effects of Spatial And Temporal Scales on Cultural Services Valuation Areas', 90(2) *Journal of Environmental Management* 1050 (2009).
[68] Pascual and Muradian, above n. 10, at 211.

Studies on forests mainly applied cost based techniques (avoided cost) to estimate regulating services, production-based methods (factor income) for provisioning services, revealed preference (TCM) for cultural services and stated preference (CVM) for supporting services.

Results tend to vary when dealing with the valuation of wetlands. In this case, stated preference methods have been used as primary approach to cultural and supporting services, probably because of specific pitfalls in revealed preference techniques, mainly scarcity of valid data.[69] Some limitations of valuation methods may be overcome by combining more than one method, the so-called 'hybrid valuation'.[70]

7. Valuing ecosystem services under uncertainty

Due to the complex nature of ecosystems, valuing services may raise several issues in addition to those specifically related to the methods illustrated above. The first critical issue in the monetary valuation of ecosystem services is uncertainty.

Uncertainty may be defined as the possibility to identify in advance all possible consequences of a decision but not their probability.[71] It needs to be distinguished from the state of risk where all possible outcomes of a decision and their probability can be enlisted in advance.[72] On the other hand, a state of 'radical uncertainty' is likely to occur where not all possible consequences can be identified before taking a decision.[73]

Scholars pointed out that valuation techniques regarding ecosystem services may be affected by uncertainties stemming from gaps in knowledge about:

- ecosystem dynamics
- social preferences
- technical issues in valuation
- insurance value and non-linear ecological changes of ecosystems
- valuation across stakeholders
- valuation in developing countries
- discounting future values

[69] Barbier, above n. 34.
[70] For more references on this method, see Pascual and Muradian, above n. 10, at 211.
[71] F.H. Knight, 'Risk, Uncertainty, and Profit', Hart, Schaffner, and Marx Prize Essays, no. 31 (1921).
[72] Ibid.
[73] R. Perman, Y. Ma, J. McGilvray & M. Common, *Natural Resource and Environmental Economics* (2003).

The following sections illustrate these sources of uncertainty in detail.

7.1 Ecosystem dynamics

The first source of uncertainty is due to the state of the art in science about the linkage between the state of ecosystems and the delivery of ecosystem services. In this case, it is straightforward that the valuation of ecosystem services varies based on the scientific information available. If the scientific evidence is not sufficiently robust to explain how specific states of the ecosystems (e.g., biodiversity) contribute to ecological functions, then it is better to apply stated preference methods of valuation to natural stocks rather than to biodiversity.[74]

However, it might also happen that probability distributions about states of nature can be objectively assigned. In this case, valuing ecosystem services is possible by weighting each potential outcome and then summing up the probability-weighted outcomes. For instance, if certain states of nature are expected with certain probabilities, related ecosystem services may be weighted accordingly.[75]

The uncertainty about the supply of ecosystem services is particularly relevant when it comes to stated preference methods of valuation. Some scholars demonstrated that uncertain future supplies of ecosystem services have an impact on option values.[76] In order to correctly measure social preferences under uncertainty, contingent valuation (CV) studies have been enriched with risk indexes which reflect individual perception of probabilities of given events. However, according to some studies, surveyed people tend to provide biased answers to questions about risk perception. For instance, the probability of negative events tends to be overestimated because objective perceptions of events' probabilities might be confounded by negative feelings of future losses.[77] Therefore, stated preference experts prefer not to base risk indexes on probabilities of supplies of ecosystem services, as this kind of information is likely to

[74] P. Nunes and J. van den Bergh, 'Economic valuation of biodiversity: Sense or nonsense?', 39(2) *Ecological Economics* 203 (2001).

[75] E.B. Barbier, 'Valuing ecosystem services as productive inputs', 22(49) *Economic Policy* 177 (2007) applies the expected damage function (EDF) – which is basically probabilistic and close to the methodologies used in risk analysis – to coastal ecosystems in order to estimate the value of flow regulation of rivers under uncertainty. Since the probability of damaging events on coastal areas (such as, storm events) can be linked to specific states of the biome (the wetland), it is possible to estimate the WTP provided that sufficient data on past damaging events and changes in ecosystems are available. The information gathered about past damage events can be used in this model to calculate the value of a wetland in terms of protection from the occurrence of damage events.

[76] The WTP for option values under uncertainty (given different future scenarios of the supply of ecosystem services) tends to change – and to be lower, where future uncertainty is reduced (Brookshire *et al.* 1983).

[77] M. Rekola and E. Pouta, 'Public Preferences For Uncertain Regeneration Cuttings: A Contingent Valuation Experiment Involving Finnish Private Forests', 7(4) *Forest Policy and Economics* 635 (2005).

undermine the quality of the valuation results. In general, very few CV studies measured values under uncertainty.[78]

7.2 Social preferences

The second issue to consider in valuing ecosystem services is related to the uncertainty of people about their social preferences. In other words, stated preference methods assume that people are aware of how much they would be willing to pay for a change in provision of ecosystem services. However, empirical studies demonstrated that respondents to surveys are uncertain about their WTP.[79] Bateman et al. explain that this is plausibly due to the heuristic way people use to process information provided in contingent valuation studies about intangible goods in hypothetical markets.[80] Acknowledging preference uncertainty brings about an adjustment to the utility function of utility-maximising individuals. In particular, point estimates of WTP should be replaced by intervals in which the true WTP lies.[81] Based on this assumption, three approaches may help dealing with preference uncertainty in stated preference valuations. The first approach is to directly ask surveyed people about their level of certainty on WTP. This may help uncover whether the attitude to valued goods or services is positively correlated with a certain level of certainty.[82] Yet, this approach does not solve uncertainty in itself.[83] The second approach is to include uncertainty in WTP questions through a polychotomous choice approach where people can express how much they would be more or less available to pay certain goods and services at the given prices.[84] Yet, it is unsure whether people interpret in the same way vague concepts, such as 'extremely unlikely' or 'extremely likely'. The third and most promising way to deal with preference uncertainty is to ask people to express ranges of values (rather than one specific value) for changes in provisions of ecosystem services.[85] Yet, it is not clear what should be the range of values.[86]

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[78] For references to these studies, see Pascual and Muradian, above n. 10, at 215.
[79] *Ex multis*, A. Alberini, K. Boyle & M. Welsh, 'Analysis Of Contingent Valuation Data With Multiple Bids And Response Options Allowing Respondents To Express Uncertainty', 45(1) *Journal of Environmental Economics and Management* 40 (2003).
[80] W.G. Hutchinson, I.J. Bateman, D. Burgess & D.I. Matthews, 'Learning Effects in Repeated Dichotomous Choice Contingent Valuation Questions', Paper No. 59, Presented at the Royal Economic Society Annual Conference (2004).
[81] J. Loomis and E. Ekstrand, 'Alternative Approaches For Incorporating Respondent Uncertainty When Estimating Willingness To Pay: The Case of The Mexican Spotted Owl', 27(1) *Ecological Economics* 29 (1998).
[82] Ibid.
[83] Pascual and Muradian, above n. 10, at 217.
[84] Alberini*,* above n. 80.
[85] N. Hanley, B. Kriström & J.F. Shogren, 'Coherent arbitrariness: On value uncertainty for environmental goods', 85 *Land Economics* 41 (2009).
[86] Pascual and Muradian, above n. 10, at 217.

7.3 Technical issues in valuation

The third source of uncertainty is caused by conceptual and methodological issues of methods that are likely to undermine the accuracy of valuation estimates.

With regard to stated preference studies, accuracy problems mainly deal with the credibility of answers, on the one hand, and the error due to the divergence between WTP and WTA, on the other hand.

About the former (credibility of answers), the crucial point of the debate is whether estimates of non-use values may be regarded as credible given that there is no other method to directly elicit these values.[87] Although the existence of an upward 'hypothetical bias' (difference between hypothetical and actual statements of value) has been rebutted in the literature,[88] it is largely recognised that the truthfulness of answers depends on the quality of the survey design and how well its structure creates incentives to reveal true preferences.[89] Additional errors in valuation might be in any case determined by the size of the sample and the nature of the good to be valued and they are expected to be fairly large.[90]

About the latter (WTP-type questions), it is well-known that answers to WTP-questions on goods are affected by large errors when compared to WTA-questions.[91] However, practitioners of contingent valuation studies defend the use of WTP for practical reasons and as a 'conservative choice'.[92]

Accuracy problems are also raised by revealed preference methods and pricing techniques. Here, the same availability of market data (or their quality), on the one hand, and the fact that estimates do not take into account non-use values, on the other hand, bring to the conclusion that these valuation methods only allow to obtain a lower bound estimate of the value of ecosystem services.[93]

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[87] Ibid.

[88] A meta-analysis that compared estimates from CV surveys and revealed methods found out that there was no statistically significant upward bias in CV studies. See R.T. Carson, N.E. Flores, K.M. Martin & J.L. Wright, 'Contingent Valuation And Revealed Preference Methodologies: Comparing The Estimates For Quasi-Public Goods', 72(1) Land Economics 80 (1996).

[89] R.T. Carson, P. Koundouri & C. Nauges, 'Arsenic Mitigation in Bangladesh: A Household Labour Market Approach', TSE Working Papers 09–106, Toulouse School of Economics, Toulouse, (2009).

[90] Pascual and Muradian, above n. 10, at 218.

[91] The divergence between WTA and WTP has been already examined in chapter II. More specifically, scholars proved the existence of an "endowment effect" (Knetsch 2005) and various empirical studies found out a large disparity between WTA and WTP for marketed goods (Kahneman *et al.* 1990). Traditionally, the ability of CV to measure consumer preferences is highly questionable. See P. Diamond, 'Testing The Internal Consistency Of Contingent Valuation Surveys', 30(3) *Journal of Environmental Economics and Management* 265 (1996).

[92] NOAA (National Oceanic and Atmospheric Administration) Report of the NOAA panel on Contingent Valuation, Federal Register 58/10 (1993).

[93] Pascual and Muradian, above n. 10, at 219.

In conclusion, technical uncertainties are likely to affect all existing valuation methods due to biases caused by the design of stated preference studies or to the quality of market data in revealed preference analyses. Therefore, it seems that accuracy in valuing ecosystem services is hardly achievable unless a data fusion approach is adopted.

Despite their residual role in the literature, data enrichment or data fusion approaches represent possible ways forward to overcome the above mentioned technical uncertainties.[94] These approaches combine data and models both from stated and revealed preference methods in a way that advantages and shortcomings can be reciprocally counterbalanced. To be more specific, combining the two approaches allows to use highly reliable (valid) data from revealed preference (e.g., estimates from hedonic price models) together with data from actual behaviours (e.g., the WTP for aesthetic benefits). In this way, the former reflect real choices and market constraints while the latter might take into account hypothetical changes in preferences determined by policy events which lie outside market data.

The advantage of data fusion approaches are thus represented by the increased amount of information, the possibility to cross-validate findings and the identification of range values.[95] The disadvantage lies in the fact that these methods can be only employed to value ecosystem services with clear direct-use values. The second disadvantage is that the role of this approach in the literature is still residual and not too many studies are available.

Alternatively, the preference calibration approach might be employed to calibrate one single preference function by comparing multiple values obtained from various valuation methods, such as hedonic pricing, travel cost and contingent valuation.[96]

7.4 Insurance value and non-linear ecological changes of ecosystems

Ecosystems hold an 'insurance value' that is dependent on their resilience, intended as the conditional probability of a regime shift from one stability state to another given the current disturbance level.[97] More simply, the ecosystem resilience is the capacity to accommodate perturbations and to maintain a certain functionality.[98] Resilience may in turn depend on systems' features, such as its functional diversity and redundancy, and the existence of critical

[94] Ibid.

[95] T.C. Haab and K.E. McConnell, Valuing Environmental and Natural Resources: The Econometrics of Non-market Valuation (2002).

[96] V.K. Smith, G. Van Houtven & S.K. Pattanayak, 'Benefit Transfer Via Preference Calibration: "Prudential Algebra" For Policy', 78 *Land Economics* 132 (2002).

[97] Pascual and Muradian, above n. 10, at 220.

[98] Ibid.

thresholds which separate regimes.[99] Ecologists have largely studied how the achievement of thresholds and subsequent shifts may change the capacity of the ecosystem to provide services in a non-linear way.[100] However, causes of regimes' shifts have not been totally unveiled, yet.[101]

Arguably, this capacity further affects the economic value of ecosystems. In other words, it is possible to establish a linkage between the state of an ecosystem, the distance from critical thresholds for a regime shift and the economic value of the ecosystem.[102] When systems are very close to thresholds, it is very difficult to predict nonlinear changes following regime shifts. As a consequence, carrying out a standard valuation under these circumstances would turn out extremely unreliable, if not impossible.[103] The reason is that standard valuation approaches to estimate the total economic value are based on marginal changes over "non-critical" states of ecosystems.[104] However, environmental economists tried to value systems' resilience as an asset, a natural capital stock which yields an 'insurance flow' of services and that can lead to certain changes of future social welfare.[105] How the value of resilience influences the economic value of ecosystem services can be derived from three considerations. First, standard economic valuation approaches measure marginal changes of values based on the linearity assumption that human disturbances produce proportional changes on the margin.[106] Secondly, current scientific knowledge about the distance of ecosystems from threshold levels is still limited. In order to enhance the predictive capacity of regimes shifts, more resources and time are needed.[107] Thirdly, real benefits from services and goods are admittedly clearer after they are totally lost.[108] To sum up, standard valuation approaches are not reliable if ecological thresholds are sufficiently close and potentially irreversible non-marginal effects of regime shifts are likely to occur. On the other hand, current studies on ecosystem dynamics are still insufficient to carry out accurate monetary valuations when ecosystems are close to thresholds.

[99] For more references on the linkage between biodiversity and ecosystem functioning, see *ibid.*, at 221.
[100] Ibid. for more references.
[101] Scholars studied how human interventions can influence the probability of a regime shift or how invasive species can lead to drastic regime shifts.
[102] K.E. Limburg, R.V. O'Neill, R. Costanza & S. Farber, 'Complex Systems and Valuation', 41(3) *Ecological Economics* 409 (2002).
[103] Ibid.
[104] Turner *et al.*, above n. 9.
[105] Pascual and Muradian, above n. 10, at 223.
[106] E.B. Barbier *et al.,* 'Coastal Ecosystem-Based Management With Nonlinear Ecological Functions And Values', 319 *Science* 321 (2008).
[107] R. Contamin and A.M. Ellison, 'Indicators of Regime Shifts in Ecological Systems: What Do We Need To Know And When Do We Need To Know It', 19(3) *Ecological Applications* 799 (2009).
[108] V. Arild & D. Bromley, 'Choices without Prices without Apologies', 26(2) *Journal of Environmental Economics and Management* 129 (1994).

More reliable valuation approaches might take account of uncertainty through the application of the precautionary principle and the possible inclusion of the value of resilience.[109]

7.5 Valuation across stakeholders

Scholars point out that identifying relevant 'stakeholders' is a crucial issue in all steps of the economic valuation of ecosystem services (e.g., for the identification of policy objectives and trade-offs in the use of ecosystem services).[110] Stakeholders can be defined as all persons, organisations and groups with interests in the way ecosystem services are used and managed.[111] Every stakeholder may have different reasons for assigning values to various ecosystem services depending on various factors, such as cultural background and impact of services on living conditions.[112] Therefore, identifying the groups and relative reasons underlying economic values may provide better information than just listing values.[113] This information can in turn be conveyed to the decision-makers in order to set policy objectives and tackle trade-offs in the use of ecosystem services. In other words, resorting to a stakeholder analysis allows to identify who is going to gain and who is going to lose from policies that determine changes of ecosystem services. Moreover, it allows to better account for the scale of the ecosystem services to be valued. In fact, ecosystem services that are likely to provide benefits to a larger set of stakeholders are necessarily valued differently compared to services whose benefits go to fewer people. Ecological economists found out that provisioning services are valued more by local stakeholders whereas regulating or cultural services tend to receive a higher value from global stakeholders.[114]

Nevertheless, stakeholders' values have not been sufficiently explored in economic valuation research until the 2000s.[115] Scholars highlighted that a stakeholder-approach requires first to prioritise stakeholders according to their role in the management of ecosystem services[116] and, secondly, to identify the groups that are able to manage future changes in the provision of ecosystem services.[117] Once all stakeholders types and roles have been correctly represented,

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[109] Pascual and Muradian, above n. 10, at 225.
[110] De Groot, above n. 57.
[111] Pascual and Muradian, above n. 10, at 227.
[112] L. Hein *et al.*, 'Spatial Scales, Stakeholders And The Valuation Of Ecosystem Services', 57 *Ecological Economics* 209 (2006).
[113] Adamowicz, above n. 54.
[114] Martín-López, above n. 55.
[115] C. Manski, 'Economic Analysis Of Social Interactions', 14 *Journal of Economic Perspectives* 115 (2000).
[116] De Groot, above n. 57.
[117] C. Fabricius *et al.*, 'Powerless Spectators, Coping Actors, And Adaptive Co-Managers: A Synthesis Of The Role Of Communities In Ecosystem Management', 12(1) *Ecology and Society* 29 (2007).

the whole valuation process should involve them through participatory tools or deliberative monetary methods.[118]

7.6 Valuation in developing countries

Despite the fact that people living in developing countries shows the highest reliance on natural resources, such as food, fuel, building material and medicines,[119] research on the economic valuation of biodiversity has been mainly conducted in developed countries. Among the few biodiversity valuation studies in developing countries, Asia, Africa and South America have been attracting attention in descending order over the past years.[120] Scholars pointed out three types of challenges that may explain this choice in gathering data.[121] First of all, methodological issues might arise due to the low level of education or language difficulties. Given these circumstances, the employment of traditional survey techniques, such as interviews and questionnaires, might turn out particularly difficult and they should be better replaced with participatory approaches in valuation.[122] The second category of issues concern practical obstacles due to the fact that people in poor countries may not be used to pay with money and, thus, they may not find money as an easy way to measure the value of goods and services. This issue can be overcome by finding alternative tools for measuring the WTP.[123] The third category of issues relates to the divergence of policies upon which surveys are normally built, such as taxation as the most common payment vehicle. Further obstacles might include extreme environmental events and the absence of local research capacity to run projects due to absent inputs from policy-makers into valuation.

For all these reasons, the application of standard approaches to valuation in developing countries should be replaced by deliberative and participatory approaches and carried out by local researchers.[124]

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[118] C. Spash, 'Deliberative Monetary Valuation (DMV): Issues In Combining Economic And Political Processes To Value Environmental Change', 63 *Ecological Economics* 690 (2007).
[119] Pascual and Muradian, above n. 10, at 229.
[120] Ibid.
[121] M. Christie *et al.,* 'An Evaluation of Economic and Non-economic Techniques for Assessing the Importance of Biodiversity to People in Developing Countries', Report to the Department for Environment, Food and Rural Affairs, UK (2008).
[122] I. Fazey et al., Livelihoods and Change in Kahua, Solomon Islands (2007).
[123] P. Rowcroft, J. Studley & K. Ward, Eliciting Forest Values and 'Cultural Loss' for Community Plantations and Nature Conservation (2004).
[124] Pascual and Muradian, above n. 10, at 231.

7.7 Discounting future values

Discounting is a key tool in environmental economics to tackle uncertainty, risks and equity.[125] It deals with how ecosystems and biodiversity should be valued today in order to take account of the effects of ecosystem losses tomorrow (to meet the future generations' needs).[126]

Environmental economists have been traditionally using a capital investment approach for most resource allocation issues.[127] In particular, they believe that natural resources should be allocated to investments with the highest rate of return, taking into account uncertainty, risk and risk attitude. So, for instance, the decision of maintaining a forest or, alternatively, cutting it down to sell timber should be mainly based on the rate of return for the money invested after selling the wood. However, from an ecological perspective, this approach – that would cause irreversible biodiversity losses, is clearly based on the view of trees merely as a form of capital of which the environmental characteristics would be fungible with economic investments.[128] As a consequence, a private investment decision based on money earned from selling timber would actually ignore the role of ecosystems in providing other, different, services.

Drawing on the example above mentioned, it is possible to imply that the discount rate is the reverse of the interest rate and, more precisely, the return on money that one would lose for preserving natural resources rather than monetizing them. The discount rate is thus based on the view of ecosystems as a source of services with direct market value (revenues from fishing, ecotourism, etc.). In other words, when calculating costs and benefits of development projects, discounting tends to emphasize more short-term economic benefits rather than the costs of lost environmental services which have a small or no monetary value.[129]

Lastly, it is worth mentioning that several environmental economists working outside the neoclassical domain came to the recent conclusion that the standard economic model – including discounting – is inadequate to analyse current environmental issues.[130] Particularly, debates over two of the most urgent issues of our times (biodiversity loss and climate change) have progressively argued against the ability of the standard economic model to capture future values of natural resources. This is basically due to the fact that economic values cannot take

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[125] TEEB (2008). An Interim Report. European Communities, at 28.

[126] Clearly, discounting has to do with intergenerational issues and the whole debate on sustainability. For a review of economic approaches to sustainability, see E. Neumayer, *Weak Versus Strong Sustainability: Exploring the Limits of Two Opposing Paradigms* (2013).

[127] J. Gowdy et al., 'Discounting, Ethics, and Options For Maintaining Biodiversity And Ecosystem Integrity', in P. Kumar (ed), TEEB, *The Economics of Ecosystems and Biodiversity Ecological and Economic Foundations*, IUCN, at 257 (2010).

[128] Ibid.

[129] See ibid. for an example based on the value of the Amazon forest.

[130] P. Dasgupta, 'Discounting Climate Change', 37 *Journal of Risk and Uncertainty* 141 (2008).

fully account of specific characteristics of ecosystems and biodiversity losses, such as the global and local scale of environmental phenomena, long-term and irreversible impacts, non-linear changes, non-linear and non-marginal changes, issues of inter and intra-generational equity.[131]

In the wake of the above, the practice of discounting, and namely to apply a negative discount rate to the environment, has been heavily criticised. First, because it refers to individuals deciding how to use natural resources at one point in time and naturally led to put more weight on present rather than future gains. However, it has been stressed that discount rates assume that a biodiversity loss will be valued much less in the future and there is no reason why future generations should be valuing their well-being in a different way than today. Secondly, discount rates neglect that the future growth rate of consumption should be negative since current generations have been basing their economic growth upon depletion of natural resources that should have been passed to future generations.

In light of this criticism, alternative economic approaches have proposed to introduce multiple discount rates according to the time, degree of uncertainty, ethical responsibilities to future generations and policy objectives. They remark that a positive discount rate boosts economic investments, on the one hand, but also environmental degradation, on the other hand.

8. Benefits transfer to value ecosystem services

Benefits transfer (BT) represents a practical way to value ecosystem services if one wants to save money and time in carrying out studies specifically addressed at valuing the interested ecosystem.[132] It allows to estimate the value of an ecosystem by transferring the already available values of a site with a similar ecosystem to the site to be valued.

8.1 Benefit transfer methods

Based on the most recent literature, BT methods for ecosystem services can be distinguished into four types with increasing complexity.

The first and simplest method is called 'unit benefit transfer' and it consists of multiplying the mean unit value of the already estimated ecosystem by the quantity of service at the site to be

[131] For more references on this point, see Gowdy, above n. 128, at 264.

[132] 'An approach to overcome the lack of system specific information in a relatively inexpensive and timely manner', quote from Pascual and Muradian, above n. 10, at 231.

valued. The mean unit value can be obtained by aggregating values expressed by the people that admittedly hold values for the ecosystem or by aggregating values over the physical area occupied by the ecosystem to value.[133] This method is apparently simpler but it might not capture accurately differences between the considered sites.

The second type is the 'adjusted unit transfer' which makes simple adjustments for differences in incomes and prices between the two sites.

The third type is the 'value function transfer' that employs functions obtained through traditional methods (travel cost, hedonic, contingent valuation, choice modelling) enriched with parameter values of the site to be valued. In this way, transferred values better reflect the characteristics of the site in object and the differences between the site to be valued and the site of which the values are transferred.

The fourth type of BT method is the 'meta-analytic function transfer' which plugs into the value function results from several studies rather than from a single study in order to include more site characteristics (socio-economic, physical, study characteristics).[134] The latter methods seem to be more accurate. However, more complexity does not necessarily lead to lower transfer errors, since the precision of benefit transfer valuations relies more on the availability of high quality primary valuation studies on sites with very similar characteristics.[135] Absent these kinds of primary data, the level of complexity tends to increase and to make benefit transfer extremely time consuming. The concept of transfer errors needs anyway to be better investigated.

8.2 Challenges in benefit transfer

Any of the above-mentioned methods is likely to pose relevant challenges for their accuracy.

First and utmost, transferred values may be affected by significant transfer errors due either to inaccuracies in primary valuation estimates (weak methodologies, unreliable data) or to the so-called generalisation errors occurring in transferring values without taking into account population or environmental differences between the sites. Before using information obtained through value transfer or before opting for BT, it is therefore worth ascertaining the scale of

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[133] Ibid. The study pinpoints the fact that it is often more practical to base the aggregation of values to be transferred on the spatial extent of ecosystems rather than on the households, since it is not always possible to identify the beneficiaries of ecosystem services.

[134] R.S. Rosenberger and T.T. Phipps, 'Correspondence And Convergence In Benefit Transfer Accuracy: A Meta-Analytic Review Of The Literature', in S. Navrud and R. Ready (eds), *Environmental Values Transfer: Issues and Methods* (2007).

[135] Pascual and Muradian, above n. 10, at 232.

potential errors. The level of acceptable error depends in turn on the specificity of the context. Higher transfer errors are admissible for regional assessments but not for the compensation of environmental damage.[136]

The second challenge in BT is the aggregation of transferred values (by multiplying unit values by the quantity of demanded or supplied service). If values are expressed per beneficiary, the aim of the aggregation is to apply the individual WTP to the relevant population. The latter is thus determined by identifying, for instance, the size of the market for the ecosystem service. If values are determined per unit area, the aggregation simply extends transferred values to the whole area of the ecosystem to be valued regardless its demand level. Aggregation may pose additional limits in double counting of values where ecosystem services are not entirely independent.[137]

The third challenge in conducting accurate BT is given by the spatial scale of ecosystem services. In fact, ecosystem services can be supplied in large or regional areas as well as on-site (recreational services from a forest) or off-site (climate regulation). Likewise, beneficiaries can be identified either locally or at a global scale. Given that, an accurate calculation of the total economic value of ecosystem services requires consideration of the spatial scale in order to take fully account of heterogeneity, substitute and complementary ecosystem services and spatial discounting. The spatial scale can be modelled through GIS.[138]

The fourth challenge is the need to make adjustment when transferring values between sites because of different characteristics (size of the ecosystem, types of ecosystem services, number of beneficiaries) and contexts (availability of substitute and complementary services). For instance, the size of vegetation in an area naturally influences the extent of flood protection of coastal areas. For this reason, it is important that BT takes into account the differences in site characteristics or groups of beneficiaries either by applying the unit transfer BT only to similar sites or by including parameters to adjust transfer value functions.[139] The availability of substitute (or complement) sites in the vicinity of an ecosystem needs to be equally considered, since it is expected to reduce (or increase) the value of ecosystem services from that ecosystem.[140]

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[136] Pascual and Muradian, above n. 10, at 233.
[137] Turner, above n. 65.
[138] I.J. Bateman, A.A. Lovett & J.S. Brainard, Applied Environmental Economics: A GIS Approach to Cost–Benefit Analysis (1996).
[139] Pascual and Muradian, above n. 10, at 236.
[140] For instance, Ghermandi found a negative relationship between the value of wetland ecosystem services and the spatial scale of wetland. See A. Ghermandi *et al*, 'Exploring Diversity: A Meta-Analysis Of Wetland Conservation And Creation', Proceedings of the 9th International BIOECON Conference on Economics and Institutions for Biodiversity Conservation (2007).

The fifth challenge in transferring values from one ecosystem to another is the possibility that ecosystem services show non-constant marginal values due to ecological relationships or decreasing marginal utility gained by beneficiaries.[141] For example, habitats may have limited value until they reach a sufficient size to support a viable population of large predators. Once this size has been reached, habitats tend to show increasing returns to scale. In light of these circumstances, BT might be improved by using estimated value elasticities with respect to the size.[142]

The sixth challenge in BT is to consider distance decay and spatial discounting when aggregating transferred values. In fact, scholars found out a negative relationship between values of ecosystem services and their distance from relative users (that is, beneficiaries tend to assign a lower value to more distant ecosystem services).[143] In particular, direct use values decline with distance according to the availability of substitute services. Conversely, non-use values show lower (or null for existence values) spatial discount rates in the spatial discounting literature.[144] In order to avoid overestimations of total values (or underestimations where estimations are limited to local households[145]), downward adjustments and spatially sensitive valuation functions might allow to obtain a more accurate WTP.

The seventh challenge in BT is the need to adjust the estimation of WTP based on differences in income levels. It is indeed quite well-known that marginal utilities in consumption are valued differently by poor and rich people (with the latter gaining less utility from an additional unit of good or service). Likewise, a decline in ecosystem services provision is expected to cause a greater welfare loss in poor countries. In order to overcome this challenge, transferred valued should be adjusted through the use of equity weights (*e.g.*, income elasticities)[146] and local data,[147] especially when applying data of developed countries to developing countries.

The eighth challenge is given by the availability of (high quality) primary valuation studies of all relevant ecosystem types, ecosystem services and contexts. Seemingly, only wetlands and forests have been largely investigated in the ecosystem services valuation literature. Also,

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[141] L.M. Brander, P. van Beukering & H.S.J. Cesar, 'The Recreational Value of Coral Reefs: A Meta-analysis', 63 *Ecological Economics* 209 (2006).

[142] Ibid.

[143] J. Bateman, A. Brainard, J. Jones & A. Lovett, 'Geographical Information Systems (GIS) as the Last/Best Hope for Benefit Function Transfer, Benefit Transfer and Valuation Databases: Are We Heading in the Right Direction?', United States Environmental Protection Agency and Environment Canada (2005).

[144] G.M. Brown, P. Reed & C.C. Harris, 'Testing A Place-Based Theory For Environmental Evaluation: An Alaska Case Study', 22(1) *Applied Geography* 49 (2002).

[145] J.B. Loomis, P. Kent, L. Strange, *et al.* 'Measuring The Total Economic Value of Restoring Ecosystem Services in an Impaired River Basin: Results From a Contingent Valuation Survey', 33 *Ecological Economics* 103 (2000).

[146] J.B. Jacobsen and N. Hanley, 'Are There Income Effects On Global Willingness To Pay For Biodiversity Conservation?', 43(2) *Environmental and Resource Economics* 137 (2008).

[147] D. Anthoff, R.J. Nicholls & R.S.J. Tol, 'Global Sea-Level Rise and Equity Weighting', Working Paper FNU-136 (2007).

recreation services seem to be more represented in the literature compared to regulating services. This gap in the available information for BT is likely to compromise the robustness of the method and to impede to scale-up values of ecosystem services across larger geographical areas.

The last and most important challenge relates to the transfer of values for entire ecosystem sites, the so-called “scaling-up”. When dealing with stocks of ecosystem services, it is not possible to estimate the total value by just adding up values of smaller ecosystem sites because of nonlinear changes in the provision of ecosystem services. Marginal values are therefore influenced by these large-scale changes and estimated value elasticities with respect to ecosystem scarcity need to be adopted.[148] However, predictions about the future demand of ecosystem services is not always easy. In particular, Farley proved that dynamics of changes of critical ecosystem services for which no substitutes are available are more difficult to estimate.[149] While changes of values for abundant stocks of natural capital show a constant tendency, the marginal value of natural capital stocks that progressively reach the ecological threshold tends to rise steeply for small changes. In other words, we tend to assign more value to small changes in the provision of ecosystem services that are reaching a level of loss from which they cannot spontaneously recover.[150] The shape of the demand curve for these types of ecosystem services is thus progressively inelastic and constant marginal values to assess changes in ecosystem supply would result in large errors of underestimation. Figure 1 on the next page shows this tendency.

[148] Brander, above n. 151.
[149] J. Farley, ‘The Role Of Prices In Conserving Critical Natural Capital’, 22(6) *Conservation Biology* 1399 (2008).
[150] Ibid.

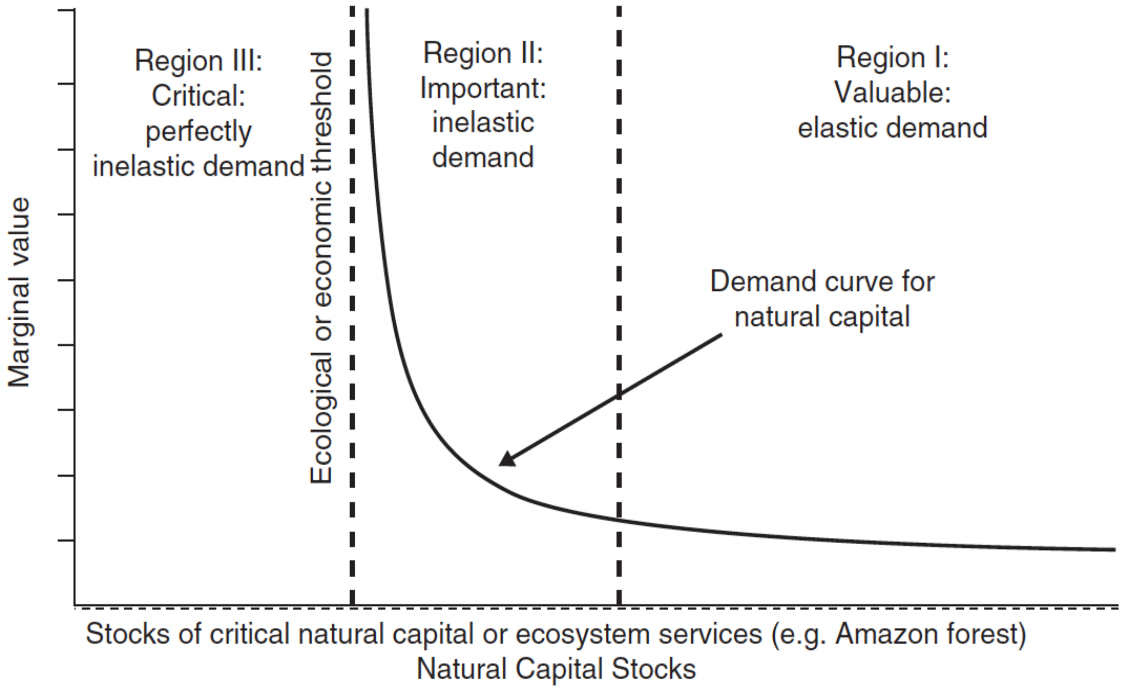

Figure 1 [The demand curve for natural capital][151]

Clearly, transferring values between ecosystems with different shapes of the demand curve might lead to big errors in valuation (risk of underestimation). Moreover, since we lack adequate scientific data about (nonlinear) changes in values of ecosystems after large-scale losses, conventional micro-economic models (TEV-based) should be better combined with alternative approaches to valuation but our knowledge about scaling-up ecosystem services values while taking into account non-constant marginal values remains very limited.[152]

9. The way forward: social network analysis and big data?

Recent trends in the economic valuation of ecosystem services are represented by the social network analysis (SNA) and the use of big data. This section wants to introduce these new streams of literature and to question whether they can offer a viable solution to assess environmental damage in litigation.

The social network analysis first emerged in the 1930s and it has been progressively employed in several domains, like physics, biology and history.[153] It allowed to shift the focus of analysis

[151] Ibid, at 1405.

[152] Pascual and Muradian, above n. 10, at 241.

[153] M. Salpeteur *et al.*, 'Networking The Environment: Social Network Analysis In Environmental Management And Local Ecological Knowledge Studies', 22 *Ecology and Society* 41 (2017).

from individuals to social categories and, more precisely, from individual behaviours to the patterns of relations among individuals. It does that by using a set of nodes to represent individuals and a set of ties to illustrate relations. The main contribution of SNA in social sciences has been to enlighten how power is distributed across the society and how social ties can explain the dynamics of the system. In the specific field of natural resources management, the SNA has offered a theoretical and methodological framework to unveil how heterogenous groups of actors interact in complex social-ecological systems and, in this way, to understand better the transmission of ecological knowledge for the management of natural resources.[154] With special regard to the economic valuation of ecosystem services, the SNA has been used to improve the estimation of cultural ecosystem services since social networks uphold relational values with cultural relevance.[155]

The second most recent tendency in the literature purports the idea that big data represent a source of environmental variables, behavioural data and, more generally, preferences that have not been yet fully explored.

To conclude, big data can indeed provide additional information which is likely to complement the results of traditional nonmarket valuation techniques, such as travel cost method and contingent valuation, while at the same time offering an alternative to benefit transfer techniques. Big data would also allow quicker assessments based on real time information. Yet, only few scholars investigated the application of big data in some case studies.

10. The ecosystem approach to the environmental damage assessment: the DWH

The last section of this chapter illustrates the application of the ecosystem services approach to a recent case of environmental damage assessment in the US, the Deepwater Horizon case (DWH). The DWH is the largest oil spill in the US history. The accident happened in April 2010 in northern Gulf of Mexico, 64 km from mainland Louisiana, with the explosion and subsequent fall of the British Petroleum's (BP) drilling platform (Deepwater Horizon), which ultimately led to the release of 200 million gallons of oil for a period of 87 days,[156] affecting

[154] *Ibid.* for more and updated references.
[155] F.M. Kilonzi and T. Ota, 'Influence Of Cultural Contexts On The Appreciation Of Different Cultural Ecosystem Services Based On Social Network Analysis', *One Ecosystem* (2019).
[156] The 1989 Exxon Valdez spilled out almost 11 million gallons out of 53 million gallons carried by the tanker. The 1979 Ixtoc 1 spill caused the release of almost 126 million gallons (Jernelöv and Lindén 1981).

1,300 miles of shoreline and coastal wetlands, an incredible number of birds, sea turtles, marine mammals, fishes, etc.[157]

In response to the accident, a major clean-up effort was implemented and approximately 1.84 million gallons of chemical dispersant were used to break up the oil into degradable oil droplets.[158] Most of the dispersant was sprayed from airplanes with possible inhalation and human health damages of workers.[159] Reportedly, to enhance the natural biodegradation of oil, an unprecedented volume of dispersants was used[160] with subsequent potential concerns about their toxicity to water organisms.[161]

Hundreds of claims and litigations were filed against BP. In October 2010, five Gulf States filed civil claims for natural resource damage and civil liability. In January 2015, a federal Court established that BP was legally responsible for the discharge of 3.19 million barrels into the Gulf for failure to perform safety tests. BP agreed to pay $ 20.8 billion in settlements (of which almost $ 9 billion environmental costs based on the restoration-cost approach) and $ 39 billion litigation costs for environmental claims.[162]

10.1 The ecosystem services approach to damage assessment

The whole process of environmental damage assessment in this case was defined 'a monumental task' because the values of all affected environmental services and goods had to be estimated and the public had to be involved in the decision making in all the affected Gulf

[157] As far as the causes of the accident were concerned, the National Commission on the BP Deepwater Horizon case conducted a 8-month review to produce recommendations and regulations. It was concluded that the technical complexity of the disaster largely exceeded industrial and regulatory safety measures. In fact, environmental agencies with supervision committed several mistakes before approving operations, such as excluding deep water drilling from regulatory requirements and carrying out inappropriately large-scale reviews. Admittedly, the Deepwater Horizon unveiled the inadequateness of technologies and regulatory responses to large oil spills. See R.L. Wallace, S. Gilbert & J.E. Reynolds, 'Improving the Integration of Restoration and Conservation in Marine and Coastal Ecosystems: Lessons from the *Deepwater Horizon* Disaster', 69(11) *BioScience* 920, at 920 (2019), citing H. Fountain, 'Lessons from the Exxon Valdez oil spill', 24 *New York Times* (2019).

[158] A.C. Bejarano, 'Critical review and analysis of aquatic toxicity data on oil spill dispersants', 37(12) *Environmental Toxicology and Chemistry* 2989 (2018).

[159] 'Although some dispersant was applied subsurface, the majority was sprayed from airplanes or surface craft onto the oil slicks. Therefore, inhalation of oil dispersant, as well as a dermal exposure of workers, was possible during remediation efforts.' See V. Castranova, 'Bioactivity of oil dispersant used in the Deepwater Horizon clean-up operation', 74(21) *Journal of toxicology and environmental health. Part A* 1367 (2011).

[160] J. Lubchenco, M.K. McNutt, G. Dreyfus, S. Murawski, P. Anastas, S. Chu & T. Hunter, 'Science in support of the Deepwater Horizon response', 109 *Proceedings of the National Academy of Sciences of the United States of America* (2012).

[161] 'The acute toxicity of dispersants is generally attributed to the disruption of biological membrane integrity, which may lead to electrolytic imbalance, loss of cell osmotic permeability, and cell lysis (National Research Council 1989, 2005; Singer et al. 1996, 1991). Dispersant toxicity has been extensively studied since the late1970s, resulting in a large body of literature' (Bejarano above n. 160, at 2990). But see also recent studies proving that dispersants are less toxic than oils (e.g., Hemmer et al. 2011; Barron et al. 2013; Claireaux et al. 2013; McConville et al. 2018).

[162] Y.G. Lee, X. Garza-Gomez & R.M. Lee, 'Ultimate Costs of the Disaster: Seven Years after the Deepwater Horizon Oil Spill', 29 *Journal of Corporate Accounting & Finance* 69, at 72 (2018).

states in order to express comments on the type of projects that they would like to see incorporated in the post-spill restoration strategies.[163] Given the complexity of the event and the potential of consequences, the US Congress asked the National Academy of Science to evaluate the impacts of the BP spill. Luckily, the first part of the task was facilitated by the large availability of data.[164]

Regarding the economic valuation of the harm, some ecological economists (Robert Costanza *et al.*) provided two monetary measurements of lost ecosystem services. The first one assumed the almost total closure of Louisiana's fishery activities and it estimated an annual loss of $ 2.5 billion. The second one calculated all values of services provided by the most affected area in the region (Mississippi River Delta) with an envisaged reduction of 10%-50% in ecosystem services and it ended up in a final total loss of $1.2–$23.5 billion per year until full ecological restoration at an indefinite time in the future and in present value (at a 3.5% discount rate).[165] In addition, some ecologists in 2016 proposed a socio-ecological approach to restoration that integrated social (economic, ethical) and ecological variables in order to achieve a successful restoration.[166] Some ecologists also pointed out that the adoption of adequate conservation beforehand would have reduced the need for extensive post-spill restoration. Other scholars proposed different estimations, such as $145 billion[167] and $2 trillion based on annual sales of coast businesses.[168]

In litigation, the US National Research Council (NRC) recommended natural resource trustees to supplement traditional assessments based on restoring equivalent resources with an 'ecosystem services approach' that could compensate also for the lost human services in the post-accident phase.[169] The final report explicitly pointed out that the ecosystem services approach differs from traditional approaches to damage assessment and to restoration, because its focus is not on the injury to the environment, but on the changes in human uses that the

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[163] B. P. Wallace, T. Brosnan, D. McLamb, T. Rowles , E. Ruder, B. Schroeder, L. Schwacke, B. Stacy, L. Sullivan, R.Takeshita & D. Wehner, 'Effects of the Deepwater Horizon Oil Spill on Protected Marine Species', in 33 *Endangered Species Research* 1 (2017).

[164] Gulf of Mexico Ecosystem Services Valuation Database maintained by Texas A&M University and the US National Research Council's (NRC) study of the ecosystem services affected by the Deepwater Horizon (NRC 2013). See C.P. Santos, C. Carollo, D. Yoskowitz, 'Gulf of Mexico Ecosystem Service Valuation Database (GecoServ): Gathering ecosystem services valuation studies to promote their inclusion in the decision-making process', 36(1) *Marine Policy* 214 (2012).

[165] R. Costanza, D. Batker, J.W. Day, R.A. Feagin, M. Martinez & J. Roman, 'The Perfect Spill: Solutions for Averting the Next Deepwater Horizon', 1 *Solutions* 17 (2010).

[166] A. Abelson, B.S. Halpern, D.C. Reed, et al., 'Upgrading Marine Ecosystem Restoration Using Ecological-Social Concepts', 66 *BioScience* 156 (2016).

[167] Y-G. Lee, X. Garza-Gomez, R.M. Lee, 'Ultimate Costs Of The Disaster: Seven Years After The Deepwater Horizon Oil Spill', in 29 *Journal of Corporate Accounting & Finance* 69 (2018).

[168] Dun and Bradstreet Bureau of Labor Statistics, 2010 Deepwater Horizon, 'Oil Spill Preliminary Business Impact Analysis for Coastal Areas in the Gulf States' (2010).

[169] Committee on the Effects of the Deepwater Horizon Mississippi Canyon-252 Oil Spill on Ecosystem Services in the Gulf of Mexico, Ocean Studies Board, Division on Earth and Life Studies, National Research Council, *An Ecosystem Services Approach to Assessing the Impacts of the Deepwater Horizon Oil Spill in the Gulf of Mexico* (2013).

environmental injury caused in addition.[170] Yet, the NRC acknowledged that the ecosystem service approach is still very early in its development and it presents many challenges, such as 'the lack of comprehensive ecosystem models'.[171] Tools of ecosystem service modelling are already in use for ex ante cost-benefit analyses or for corporate sustainability, but not within the NRDA process.[172] Therefore, based on the current state of scientific knowledge and ecological data available, it can supplement but not replace traditional ways to value environmental damage.

10.2 The ecosystem services approach to wetland valuation

Even if very early in its development, the ecosystem services approach might still provide useful information for post-accident restoration.

In the BP post-spill strategy, some ecologists proposed to better link the post-spill restoration with the previous restoration policies in the Gulf of Mexico that took into account wetland functions.[173]

Traditionally, wetlands have been seen as 'wastelands' with potential harmful effects on human health. For this reason, it became a widespread practice to drain them extensively and convert for extensive agriculture, fish ponds, industrial or residential land. The advancement of scientific knowledge over the past decades has instead proved that wetlands can provide plenty of valuable functions, such as flood alleviation, pollutants retention, groundwater recharge (regulatory functions), provision of fish, fuelwood, timber, sediments for agriculture, recreation for tourists, but also biodiversity, cultural heritage and amenity. As a consequence, an inverse trend of wetland conservation started in several countries. It can be seen by the number of policies implemented to halt wetland degradation and loss, the political recommendations to use wetlands in sustainable manners and the initiatives launched to quantify better wetland values. For instance, the Ramsar Convention on Wetlands,[174] the Convention on Biological Diversity, the UN Commission on Sustainable Development, OECD, IUCN, Wetlands International and WWF have been promoting now for three decades

[170] Ibid., at 1.
[171] Ibid., at 17.
[172] This is confirmed by C.A. Jones and L. DiPinto, 'The Role of Ecosystem Services in USA Natural Resource Liability Litigation', 29 *Ecosystem Services* 333, at 346 (2018).
[173] E.B. Barbier, 'Coastal Wetland Restoration and the "Deepwater Horizon" Oil Spill', 64(6) *Vanderbilt Law Review* 1819 (2011), at 1823ss.
[174] The Ramsar Convention was created precisely to promote the conservation of wetlands by emphasizing that many ecological services have value even if they are not traded in the market.

research, analysis and communication of more accurate information on the economic values of natural resources and wetlands in order to improve the decision making. In 1997, Barbier, Acreman and Knowler compiled a first review of techniques and examples of wetland valuation in order to complement traditional techniques limited to market prices of products such as fish and timber.[175] To achieve the goal of promoting more sustainable uses of wetlands, the authors followed a tripartite analysis.

First, they showed the real value of wetlands by emphasizing their biological components, their chemical functions and their attributes in terms of biodiversity.

Secondly, they illustrated the new methods of economic valuation promoted by the Ramsar Convention to help decision-makers to measure all benefits of wetlands and adopt more sustainable policy decisions.

Thirdly, they provided a framework to analyse the net economic benefits of wetlands depending on their nature, use and geographical area, but also on political, social, historical and economic factors. An alternative methodology is provided where rare species are at risk. In this way they did not only prove the importance of interdisciplinary (ecological and economic) approaches for a correct valuation, but they also showed how the choice of the appropriate valuation technique depends on the ultimate goal of the policy and the planned wetland use (‘valuation should not be conceived of as an end in itself, but needs to be directed towards some policy issue’).[176]

10.3 The challenge of the equivalency analysis for wetlands

As a consequence of the extensive studies on wetlands, ecologists raised criticism on the principle of ecological equivalence underneath compensatory restoration in the BP oil spill. The main point of criticism was that little attention was paid to site location within the surrounding landscape, natural patterns of plant communities, wetland hydrological regimes and long-term ecological functions.[177] In other words, long-term economic benefits resulting from restored natural resources were not taken into account but for the actual costs of restoration of equivalent ecological functions. The principle of ecological equivalence is the

[175] E. Barbier, M. Acreman and D. Knowler, *Economic valuation of wetlands: a guide for policy makers and planners*, Ramsar Convention Bureau (1997). They explicitly stated that ‘wetlands are among the Earth’s most productive ecosystems’, also ‘the kidneys of the landscape’ because of their nutrient cycle, or ‘biological supermarkets’ because they provide food, water and biodiversity.
[176] *Ibid.*, at x.
[177] Ibid.

pillar of the Habitat Equivalency Analysis (HEA) and currently the main method of environmental damage assessment under US and EU liability laws (see chapter V). It refers to the possibility of compensating the public for past losses of habitat resources through replacement projects that provide resources with ecological values equivalent to those that have been lost.[178] The HEA approach is undoubtedly beneficial in terms of time and money, since it places restoration straight at the beginning of the natural resource damage assessment, hence accelerating both compensation and restoration, minimising litigation costs and avoiding the costs of valuation studies. Yet, three main disadvantages of HEA may occur.[179] First of all, HEA might misrepresent the ecological services of wetlands. Secondly, it might lead to wrong estimates of costs and benefits. Thirdly, some wetland ecological services might be oversupplied in the long run. Therefore, in order to ensure that wetland restoration achieves both goals of compensation and restoration in a cost-effective manner (including the minimisation of costs to the environment), the state of scientific knowledge on costs and benefits of coastal wetland restoration needs to be carefully considered when restoring wetlands.

11. Conclusions

The aim of this chapter was to shed a light upon the economic valuation of ecosystem services by reviewing the main literature on methods, existing challenges, possible ways forward, and by examining a case of application of the ecosystem service approach to damage assessment.
The first conclusion is that the economic valuation of ecosystem services is currently built on conventional environmental economics methods and it presents the same challenges that have already been examined in chapter II, including uncertainties about social preferences and other technical issues. Also, none of the techniques available can provide reliable estimates if they do not take into full account the ecosystems dynamics, ecological thresholds and regime shifts. Therefore, while a full and correct valuation of ecosystem services would be needed to avoid their under-compensation (caused by the absence of market transactions and the information failure), the costs of the adjustments required to avoid errors need to be traded off with the added benefits of this method.
The second class of conclusions can be inferred from the analysis of the BP accident:

[178] Barbier, above n. 177, at 1822.
[179] Ibid.

i) ecosystem service modelling tools are not required under US law for post-damage assessments;
ii) higher standards for admissibility of ‘novel scientific evidence’ in Courts seem to be the main reason why the judiciary may be reluctant to adopt the ecosystem service approach;[180]
iii) the value added by the ecosystem service approach to damage assessment is still lower relative to the additional costs (most models of ecosystem services miss data on the environmental impacts at a very fine scale while qualitative indicators are available at lower costs);
iv) the costs for collecting data to implement the ecosystem service approach during the process of damage assessment are high and they would delay the time needed to achieve full restoration.

To summarise, there is a widespread belief in ecological economics that the ecosystem service valuation provides useful information on social preferences that should not be ignored if we want to avoid massive losses of environmental values.[181] However, further research needs to address data gaps, patterns of non-linearity and ecological modelling. New trends in ecological studies – mainly, the social network analysis and big data – will also contribute to improve the accuracy and practical viability of the economic valuation of ecosystem services.

Despite the above, it seems to the author that the application of the ecosystem services approach to the environmental damage assessment may still contribute to the ecological and the economic goals of liability in two crucial areas.

The first is the damage to key ecosystems (forests and wetlands). The economic values of their ecosystem services have been already largely investigated and mainstreamed through databases. These data are regularly updated and they may provide judges with easy-to-read information.

The second is the current practice of post-accident restoration which may benefit of an ecosystem services approach (like in the BP case). The ecosystem services approach to restoration would improve the application of the principle of ecological equivalence for compensatory restoration.

[180] With the words of Jones and DiPinto: ‘at this time, the level of uncertainty in the estimates is perceived to be high relative to the admissibility standard, particularly for such a significant case as Deepwater Horizon’. Jones and DiPinto, above n.173, at 346.

[181] Kontoleon, Pascual & Swanson, above n. 49.

Take-aways from chapter VII and bridge to chapter VIII

- The economic valuation of ecosystem services is built on conventional environmental economics methods. It presents well-known and peculiar challenges.
- While a full and correct valuation of ecosystem services would be needed to avoid under-compensation and under-deterrence, the state of the art on the ecosystem services valuation is still limited.
- The ecosystem services valuation of wetlands and forests may provide judges with easy-to-read information.
- The ecosystem services approach to restoration would improve the application of the principle of ecological equivalence for compensatory restoration.

Take-aways from Part II (review)

(1)

- Environmental damage beyond property devaluation and clean-up costs and including non-use values ('pure environmental damage') is theoretically compensable under customary international law.
- Very few decisions have been adopted by the ICJ on environmental compensation and they are neither precise nor transparent on the method of damage calculation.
- There are no clear guidelines on the method of nature valuation and the practice is therefore quite heterogenous.
- Investment tribunals seem to be more transparent and open to novel valuation methods compared to the International Court of Justice.
- Non-use losses (pure environmental damage) and interim losses were assessed with more accuracy by the UNCC which, reportedly, is a model for cost-effective and accurate environmental damage assessments at the international level.
- International conventions governing oil spills do not recognize liability for non-use losses and interim losses of natural resources.
- Both in the US and the EU, restoration has become the most common remedy because it is considered easier and cheaper and it ensures that the environment is returned to previous conditions.
- In the US, polluters are liable for a well-defined list of removal costs, interim losses pending recovery and costs of assessment.
- The environmental damage assessment in the US often relies on the HEA and it follows guidelines based on thirty years of experience.
- In the EU, a number of issues might reduce the likelihood to achieve both optimal deterrence and cost-effective restoration.
- (On compensatory restoration) neither the wording of the ELD is detailed and precise as in the US nor the economic valuation is made mandatory.
- The US law on natural resource damage assessment seems to be better placed to achieve cost-effective restoration and optimal deterrence compared to the EU Directive on Environmental Liability that was modeled after the former.
- Current liability laws do not expose polluters to the full cost of environmental accidents, including the pure environmental damage beyond clean-up and restoration costs.
- The long-term environmental impact of clean-up may not be considered by insurers, polluters and public administrations.

Take-aways from Part II (review)

(2)

- Claims for non-use values of nature are the less frequently assessed and compensated share of damage under any liability regime, except for the US Oil Pollution Act.
- Lengthy lawsuits decrease the likelihood to fully internalise the environmental costs of accidents, whereas settlements for the reimbursement of clean-up costs can make the internalization more efficient.
- Post-spill monitoring is under-supplied but crucial for optimal deterrence and cost-effective restoration.
- The economic valuation of ecosystem services is built on conventional environmental economics methods. It presents well-known and peculiar challenges.
- While a full and correct valuation of ecosystem services would be needed to avoid under-compensation and under-deterrence, the state of the art on the ecosystem services valuation is still limited.
- The ecosystem services valuation of wetlands and forests may provide judges with easy-to-read information.
- The ecosystem services approach to restoration would improve the application of the principle of ecological equivalence for compensatory restoration.
- Claims for monetary compensation of climate change damage vary across countries.
- In the US claims for monetary compensation of climate change usually relate to economic losses and costs of adaptation.
- In a few countries (Brazil and Indonesia) some plaintiffs added a specific claim for ‘climate damage’ intended as the costs of carbon release after environmental accidents.
- In Brazil, the notion of ‘climate damage’ was created by the judges and its calculation is based on the Social Cost of Carbon (SCC).
- In Indonesia, the compensation of ‘climate damage’ is in the law and it also includes the cost of carbon reduction due to the reduced capacity of burned trees to absorb CO2 (cost to restore carbon sinks). Its calculation is also based on the SCC.
- The choice of remedies for climate change liability has been sometimes influenced by arguments related to climate change and climate adaptation programmes.

PART III

How can remedies for environmental harm be improved to induce more efficient deterrence and cost-effective restoration?

CHAPTER IX

Conclusions

The overarching research question of this dissertation is whether remedies for environmental liability are providing polluters with optimal care incentives to minimise the environmental costs of accidents while, at the same time, ensuring cost-effective restoration.

This question has been selected because it encompasses both the economic and the ecological perspectives on remedies for environmental harm. The ecological approach is currently the dominant one in environmental liability laws.

From an economic perspective, the objective of environmental remedies is to provide polluters with optimal deterrence incentives by exposing them to the full social costs of their harmful activities. For a methodology of environmental damage assessment that can incentivise optimal deterrence, I mean a methodology that can induce a polluter to 'optimally' invest ex ante in order to internalise the full social costs of accidents (costs of care and expected liability).[1]

From an ecological perspective, the objective of environmental remedies is to restore the environment to its state prior to the occurrence of harmful activities. This approach does not prioritise deterrence. However, from an economic perspective, restoration should be conducted in a cost-effective manner. For a methodology of environmental damage assessment that can incentivise cost-effective restoration, I mean a methodology that can induce a polluter to 'optimally' invest in restoration in order to internalise the full costs of restoration (clean-up costs, interim losses and long-term restoration costs).[2]This general question has been split into three sub-questions for each part of this dissertation:

1. Do current methodologies of environmental damage assessment induce both optimal deterrence *and* cost-effective restoration? (part I)
2. Are remedies in the legislation and the case law inducing optimal deterrence and cost-effective restoration? (part II)
3. How can remedies for environmental harm be improved to induce more efficient deterrence and cost-effective restoration? (part III)

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[1] Polluters receive optimal incentives of care if they invest ex-ante up to the point where the marginal costs of risk reduction (or precaution) equals the marginal benefits (avoided loss or expected liability). See Chapter II, § 3.

[2] Polluters receive optimal incentives of restoration if they invest in restoration up to the point where the marginal costs are equal to the marginal benefits (i.e., the environmental benefits). See Chapter III, § 3ss.

Part I

Do current methodologies of environmental damage assessment induce both optimal deterrence and cost-effective restoration?

Part I addressed the first sub-question by comparing the traditional monetary compensation for environmental harm (obligation to pay) with the more recent restoration-based compensation (obligation to restore). This order has been clearly inspired by the evolution of liability laws at the international and national level in the European Union and the United States.

Chapter II examined the question with limited attention to the deterrent effect of conventional methods of valuation in environmental economics. The decision to limit the scope of the analysis of this first chapter to the domain of environmental economics is motivated by the fact that this scholarship exerted a profound influence on the early adoption of environmental liability laws in the US and subsequently in the EU, as well as on the judicial practice.[3] Moreover, these approaches are solely focused on monetizing the environment, which precludes any inferences regarding their ecological objectives (optimal restoration).

By evaluating strengths and limitations of traditional nature valuation methods, the chapter demonstrated that:

- there is no single approach to environmental damage assessment that can be universally applied to all forms of damage (no 'one size fits all' method) in view of optimal deterrence;
- three dimensions may be considered in order to estimate the efficiency of methods. These are: accuracy of estimates, captured value of nature and assessment costs;
- in the event of an environmental damage that is characterised by a major loss in use values, market-based or revealed preference approaches are more better placed to achieve optimal deterrence, provided that the necessary data (market prices) are readily available. This is because these approaches only encompass the use value of natural resources. As a consequence, potential polluters can easily retrieve this type of information and, thus, foresee their expected liability;

[3] Moreover, the experience of judges applying non-market valuation methods played a pivotal role in determining the shift towards a restoration-based compensation regime in liability laws in both the US and the EU.

- if, instead, the proportion of lost non-use values in relation to the total magnitude of environmental damage is considerable, market-based and revealed preference approaches should be eschewed since they fail to encompass non-use values and this would result in an underdeterrent effect for the polluter. It is more socially desirable that stated preference approaches be employed in order to achieve better deterrence, particularly in the case of unique, irreplaceable and irrecoverable natural resources, where the non-use component of the total environmental value (TEV) is significant;
- however, it has been noted that the efficiency of methods also depends on the accuracy of their outcomes and the assessment costs. These two aspects are of particular importance in the context of stated preference approaches, where greater accuracy can be achieved at the expense of higher costs of assessment. Therefore, stated preference approaches may be efficient where a) they are rigorously applied, and b) the non-use component of value is sufficiently high to outweigh the assessment costs.

In conclusion, it appears that in theory the methodologies to monetise the value of nature in environmental economics can achieve optimal deterrence by exposing polluters to the full social costs of accidents. This is true to the extent that the methodology employed is in line with the specific characteristics of the injured environment (proportion between use and non-use values), on the one hand, and the availability of data and resources to conduct the assessment, on the other hand.

Nevertheless, a number of issues in practice demonstrated that the deterrent effect of monetary compensation based on the conventional methods described above was insufficient. Primarily, assessing the monetary value of non-use losses was deemed too costly and difficult for both claimants and courts. Secondly, none of the conventional methods in environmental economics could guarantee the effective pursuit of environmental restoration (the ecological goal of liability laws). Consequently, a tendency emerged in the law to move away from the imposition of damages and to impose restoration of the damaged environment as the primary remedy. This shift occurred first in the US and later in the EU.

However, despite the significant change in legal remedies for environmental damage, scholars did not devote sufficient attention to the deterrent effect of restoration. From an economic perspective, being restoration a cost, it may be able to expose polluters to the social costs of accidents and it could thus lead to cost internalisation. Another advantage is that if polluters are aware of restoration costs, courts should not necessarily know the exact costs and an order of restoration may suffice. By reducing the costs associated with litigation, the total social costs of accidents are minimised. Yet, the efficiency of restoration is contingent upon a number of factors, including the extent to which polluters are aware ex ante of how much they have to pay as a result of their harmful activities. This point has not been subjected to deep examination in the existing scholarship of law and economics.

Chapter III aimed to fill in this research gap by offering a novel and comprehensive perspective on the obligation to restore. More precisely, it inquired whether restoration can induce both optimal deterrence *and* cost-effective remediation *in comparison with* monetary compensation. Its main finding is that, although restoration appears to be a more effective approach to achieving the ecological objective of liability (returning the environment to its previous conditions), it is constrained by several limitations that can undermine both its economic and ecological objectives.

Some of these limitations are peculiar to this remedy (complex scenarios of environmental injury, technical, ecological and financial impossibility to restore, missing incentives for compliance, time between the event and the implementation of the duty to restore), while others are in common with the monetary compensation. Regarding the former, it has been demonstrated that restoration can be an effective solution in relatively straightforward cases, where the extent of environmental harm can be quantified with ease and the obligation to restore can be clearly defined. However, it is more challenging to assess the damage and to determine the obligation to restore when the harm occurs in complex scenarios, such as a wetland providing multiple environmental services.

Furthermore, in numerous cases a real restoration of the injured environment may be impossible due to a lack of data on the baseline or the absence of suitable measurement tools (technical impossibility). Additionally, the injury may have affected protected species, vulnerable sites, biodiversity hotspots, and key ecosystems that are inherently irreplaceable (ecological impossibility). Finally, the costs of data collection and environmental restoration may be prohibitively high (financial impossibility).

Nevertheless, even in instances where the obligation to restore can be clearly defined, there may be a lack of incentives for compliance and operators may be able to influence the interests of public authorities to accept suboptimal restoration levels.

It is also possible that restoration is possible and the incentives for compliance are optimal. Despite this, there is a chance that part of the accidents' social costs may not be fully internalised by liable operators. This is particularly relevant with regard to the interim losses between pollution and full recovery, as well as lost non-use values.
Finally, the passing of time after the pollution event initiates a process of natural regeneration, which may result in reduced restoration costs at the time of the judicial decision and/or overconfident biases of liable operators.

Other limitations are in common with the obligation to pay for environmental harm. These relate to possible uncertainties, high assessment costs, low probability of detection and insolvency risks. For instance, analogous to the inherent uncertainty surrounding monetary compensation, uncertainties may arise about the target of restoration, the extent of restoration required or whether restoration should be focused on the use and/or non-use values of nature, or even on the practice of restoration. Furthermore, decisions regarding restoration are contingent upon factors such as the availability of manpower, equipment, and scientific knowledge. When clean-up is conducted by public authorities (a common occurrence), the information related to these issues is not readily available to polluters and the costs of acquiring it in advance may be prohibitively high.

Moreover, both remedies may suffer from high assessment costs. It was already mentioned that assessing damages in environmental cases can be very costly. But also the costs of restoration may be considerably high due to the uncertainties listed above, to longterm monitoring costs and to possible private interests of agencies, insurers and local communities. High assessment costs may be overcome by employing algorhythm in litigation but decisions on restoration seem to be too complex for being automated.

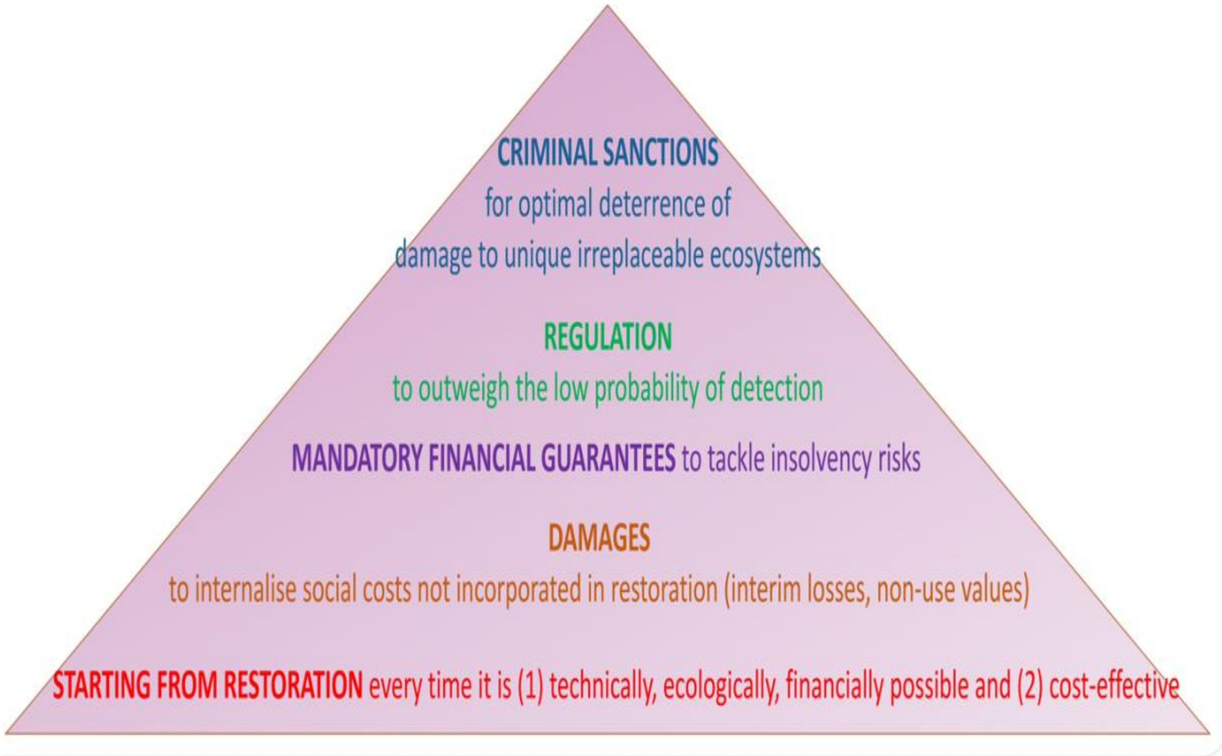
Another common problem is the low probability of detection. When the probability of the polluter being detected is lower than 1, the expected liability should be higher than the potential benefits to the polluter in order to reach deterrence. For instance, punitive damages would be needed. This is a problem both for restoration as well as for damages and it points to the need of having ex ante regulation.

A last common problem to both remedies is the insolvency risk. The assumption that potential polluters will be deterred by either the threat of a restoration or the payment of damages supposes that potential polluters are able to pay the restoration costs or the damages. In case of restoration costs or damages being higher than the polluter's wealth, liability should be accompanied with mandatory solvency guarantees, such as liability insurance.

All the afore-mentioned issues lead to possible scenarios where polluters are not fully aware of the future costs of restoration and/or they are not exposed to the full costs of environmental accidents. As a result, they may lack sufficient incentives to invest in preventing environmental harm (suboptimal deterrence). Moreover, costeffective restoration may not be achieved.

Building on the findings of Chapters II and III, it can be concluded that the answer to the first subquestion (i.e. whether current methodologies of environmental damage assessment induce both optimal deterrence and cost-effective restoration) is negative. The obligation to pay following traditional non-market valuation methods does not provide an incentive for optimal deterrence. Furthermore, the potential for achieving cost-effective restoration through traditional non-market valuation methods is uncertain. Similarly, the mere obligation to restore does not provide an incentive for optimal deterrence. Moreover, achieving cost-effective restoration is very difficult or even impossible in numerous cases.

It seems that a solution cannot be found in one single method of environmental damage assessment, neither in the mere obligation to pay nor in the mere obligation to restore. Rather, it could be found in what has been here termed an 'optimal mix of remedies for environmental harm'. The proposed model combines the remedy of restoration with damages, regulation, financial guarantees and criminal law, with the objective of increasing the likelihood of achieving both the economic goal of liability laws (optimal deterrence) and their ecological goal (cost-effective restoration), while taking into account the specificities of the injured ecosystems. The figure on the next page illustrates the optimal order of remedies starting from restoration and ending with criminal sanctions.



The case of irreversible environmental damage (i.e., the harm to unique and irreplaceable natural resources) may be particularly illustrative. In such instances, neither the obligation to pay nor the obligation to restore can ensure the combined pursuit of optimal deterrence and cost-effective restoration. Indeed, the obligation to pay might not encompass its entire value and the obligation to restore might not be technically or economically viable. In such instances, it is evident that other remedies, such as criminal sanctions and insurance, can provide more optimal incentives of deterrence (economic goal), while the concurrent utilisation of a restoration-based compensation facilitates restoration efforts to the possible extent (ecological goal).

Part II

Are remedies in the legislation and the case law inducing optimal deterrence and cost-effective restoration?

While Part I of this dissertation considered the theoretical efficiency of remedies, Part II investigated the efficiency of remedies for environmental harm in practice by examining the relevant legislation and selected cases. In line with Part I, the objective of Part II was to determine whether methods of environmental damage assessment in the law (in text and in action) are efficient. This means that they expose polluters to the total costs of their harmful activities and they allow for cost-effective restoration. The empirical multilevel and comparative analysis commenced at the international level (chapter IV) and subsequently proceeded to the national level (chapter V). Chapter VI continued with a more in-depth examination of accidents (oil spills) where international, regional and national laws overlap. Chapter VII explored the novel ecosystem approach to damage assessment that has been recently employed in the US. Chapter VIII concluded with the analysis of the monetary compensation for damage caused by climate change.

At the international level, the principle of 'full reparation' of damage has been consistently reaffirmed by the Permanent Court of International Justice and the International Court of Justice. However, when it comes to damage valuation, the decisions of the International Court of Justice on compensation for environmental damage in 2018 and 2022 are somewhat disappointing in several respects. In principle, the Court explicitly embraced a broad notion of environmental damage in accordance with the principle of full reparation, including the loss of natural resources without commercial value (beyond property losses and clean-up costs). However, the final damage award did not correspond to this conceptualisation and the Court opted for an overall assessment that was much below the sum of the single heads of damage. Even when the Court appointed an independent expert (Congo v. Uganda), the calculation of the monetary award was not transparent and clear guidelines on how to value environmental damages are missing. Conversely, some decisions of investment tribunals have been more inclined to accept new valuation methods and to have independent valuations. Also, one decision of the Inter-American Court of Human Rights was particularly interesting because it granted both reparation and compensation for pecuniary and non-pecuniary damages linked to environmental harm. In this context of considerable diversity, the United Nations Compensation Commission (UNCC) created after the 1990 Gulf War stands out for its broad notion of environmental damage, the heads of damage that could be compensated, the expertise of the technical body, the speed and cost-effectiveness of the environmental damage compensation procedure. It is noteworthy that the 'pure environmental damage', i.e. the loss of natural resources devoid of commercial value, was incorporated in the final compensation of environmental claims, in addition to the costs of restoration and all the temporary losses directly caused by illegal acts. To conclude, chapter IV demonstrated that current remedies for environmental harm at the international level are inadequate for optimal deterrence as polluters are not exposed to the full costs of accidents, except for the UNCC and investment tribunals.

Chapter V continued the previous analysis of environmental liability laws, moving from the international to the national level. It provided a detailed picture of the law on environmental damage assessment in the US and the EU. Of particular interest is the evolution of US natural resource damage assessment law, which shows how the focus shifted from developing a suitable method for monetarily quantifying the total economic value of nature to developing cost-effective restoration plans that provide equivalent services. As Jones and DiPinto (2018) observe, the 'restoration-based' compensation has become the primary remedy and 'compensatory restoration' is specifically directed towards interim losses and irreversible damage. During the 1990s, the results of this long evolution were transplanted to the EU, leading to the adoption of the EU Directive on Environmental Liability (ELD) in 2004. This proposed to remedy (and prevent) environmental damage with a set of restoration actions modelled after US laws. However, the ELD provided less detailed guidance than the US law on natural resource damage assessment. Furthermore, a number of obstacles may prevent the achievement of full restoration even when the damage is reversible. These include information costs for polluters, a lack of guidelines on primary and compensatory restoration, the impossibility of identifying liable parties and a lack of time constraints in litigation. Lastly, technical uncertainties in the EU are more pronounced due to the lack of scholarship in environmental economics and the absence of experts capable of conducting accurate equivalency analyses compared with the US. In conclusion, there are several reasons to doubt that the obligation to restore introduced by the ELD can result in polluters being held fully liable for the full costs of restoration and that it can induce optimal deterrence and cost-effective restoration.

Chapter VI examined the extent to which polluters in real cases are exposed to the full costs of accidents, with a particular focus on the compensation of environmental damage beyond clean-up and restoration costs. This includes the non-use values of nature and interim losses. The analysis concentrated on large marine oil spills, given the availability of more data and scholarship on this topic. Additionally, it is widely acknowledged that international conventions governing oil spills do not typically expose polluters to liability for non-use losses and interim losses of nature. It is therefore of interest to examine the role of national liability laws in addition to the international legal framework for the compensation of oil spill damages. In the previous chapters, the question of whether the law is providing optimal incentives to achieve both restoration and deterrence in an efficient manner was addressed. Chapter VI went even further by identifying additional aspects that equally matter for optimal deterrence and cost-effective restoration. These include optimal decisions on clean-up, incentives to claim compensation for 'pure environmental damage', length of liability lawsuits, post-accident monitoring and financial guarantees. With regard to the matter of cleanup, it would appear that the long-term environmental impact of emergency cleaning activities has been frequently overlooked by insurers, polluters and public administrations. With regard to claims for non-use values of nature, some scholars have demonstrated that they were the least frequently assessed and compensated share of damage under any liability regime (with the exception of the US Oil Pollution Act). With regard to the duration of litigation, this has been consistently lengthy, to the extent that the probability of internalising the full cost of restoration may be diminished, thereby encouraging overoptimistic biases. Furthermore, post-spill monitoring has been under-supplied, despite its crucial role in facilitating full restoration. Finally, financial guarantees required by the international conventions governing oil spills have primarily addressed clean-up costs, which raises questions about their adequacy. This implies that even when restoration occurs, polluters may not be held fully accountable for the social costs of accidents. Indeed, the mere obligation to restore the environment or to pay a certain amount of money may not be sufficient to achieve both the ecological and the economic goal of remedies for environmental harm. The joint goal of efficient internalisation of social costs and cost-effective restoration may only be achieved through a combination of remedies providing optimal incentives (also) to clean-up, to claim compensation for pure environmental damage, to adopt judicial decisions within reasonable timeframes and to conduct post-accident monitoring on restoration. Lastly, even when liability laws provide optimal incentives for all these points, it may not be sufficient in the absence of financial guarantees for environmental damage.

Chapter VII presented a novel approach to non-market valuation proposed by ecological economists and recently applied in a case of environmental damage assessment. In contrast to traditional economists, who draw on an utilitarian anthropocentric perspective that monetises natural resources in view of their use and transformation, ecologists traditionally adopt a biocentric perspective, which rejects the commodification of the environment and argues in favour of its conservation. Nevertheless, the failure of conservationism in the 1970s, coupled with the mounting demand of the economic system for natural capital, prompted some 'ecological economists' to propose a novel approach that could, in theory, overcome the separation between conservation and development while pursuing 'conservation for development' (Folke 2006). The so-called 'ecosystem services approach' was introduced to emphasise environmental benefits that had traditionally been overlooked in environmental economics. Nevertheless, valuation frameworks of ecosystem services have been systematised according to conventional methods. This chapter therefore set out to demonstrate how traditional valuation techniques have been applied to ecosystem services and to identify the challenges that have been raised. In particular, it is evident that many uncertainties in the valuation of ecosystem services still occur given the current state of the art in ecology. While a comprehensive and accurate valuation of ecosystem services would be necessary to prevent under-compensation and under-deterrence, the current state of the art on ecosystem services valuation is still quite limited and may be of little assistance in litigation. An exception to this may be represented by wetlands and forests, for which more data and economic values are available in the Ecosystem Service Valuation Database (ESVD). In light of the limitations of the ecosystem services approach in ensuring socially desirable outcomes in liability cases, it is unlikely that this approach will be able to provide the necessary assurances until more extensive ecosystem models and a greater quantity of ecological data are available. The case of the Deepwater Horizon (BP) was finally reported as a first attempt to apply the ecosystem services approach to damage assessment with limited success. It demonstrated that the application of the ecosystem services approach to restoration would specifically improve the precision of the principle of ecological equivalence for compensatory restoration of wetlands.

Finally, Chapter VIII analysed a special category of cases in which claimants seek monetary compensation for climate-related damage. The number of climate-related lawsuits in national, regional and international courts is growing at an unprecedented rate. While most of these cases are aimed at declaring the failure of states and corporations to mitigate CO2 emissions, a small number of cases have recently been filed to seek monetary compensation for climate change damages. The aim of this chapter was to shed some light on the calculation of damages in climate litigation and to see to what extent it reflects the state of the art in climate economics. Indeed, the calculation of climate change damages has not been sufficiently explored either in climate change litigation scholarship or in law and economics. The chapter addressed this objective in three successive steps. First, climate change was presented from the perspective of a natural scientist in order to provide a scientifically grounded view of the phenomenon. Second, it summarised existing methodologies for translating all impacts into social costs. Thirdly, the existing practice of tort cases involving claims for monetary compensation was analysed. The analysis was based on cases available in the Climate Change Litigation Database of the Sabin Center at Columbia University. An interesting variety of climate change damages and valuation methods emerged, ranging from the costs of climate adaptation to the costs of restoring carbon sinks. More specifically, in the US case law, monetary compensation for climate-related damages tends to focus on economic losses (pecuniary damages) and adaptation costs. Conversely, very few cases were recorded in the non-US case law and the practice was quite heterogeneous. Interestingly, only in Brazil and Indonesia did claimants seek compensation for 'climate damage' as a separate head of damage caused by environmental accidents. Courts have calculated these damages in terms of the social cost of carbon (SCC). The SCC appears to increase the accuracy of the overall level of liability and thus the likelihood of optimal deterrence incentives. However, the current state of climate economics does not allow us to conclude that the SCC incorporates the full costs of climate change to the environment, particularly with respect to non-economic losses. Moreover, there are a number of uncertainties in the calculation of the SCC that may increase its imprecision and should be taken into account by courts. This leads to the observation that tort based climate litigation seeking monetary compensation may not provide optimal deterrence incentives to polluters if climate change damages are not assessed in a way that the level of liability is approximately the same as the damage caused by climate change.

The empirical and multi-level analysis conducted in Part II has shown that current remedies for environmental damage at the international and national level may not be optimal in terms of deterrence because polluters are not fully exposed to the full costs of accidents, especially those costs to the environment that are not financially quantifiable and for which existing valuation methods are inadequate. There is also a lack of certainty that the restoration of the damaged environment will be carried out in a cost-effective manner. These conclusions lead to the proposal of the following policy recommendations.

Part III

How can remedies for environmental harm be improved to induce more efficient deterrence and cost-effective restoration?

The analysis of the law and the case law conducted in Part II of this dissertation confirmed that, as theoretically anticipated in Part I, the way in which the environmental damage is assessed does not correspond to the economic starting points. Both at the international and domestic level, there is too much focus on restoration, which it limited to compensation for clean-up costs, rather than compensation for the total economic value of the environment.[4] As a result, potential polluters may not be fully exposed to the costs of their polluting activities ex ante, leading to underdeterrence and inadequate remediation. For these reasons, it is crucial to further incorporate economic insights into a smart design of remedies in environmental law. In order to operationalise the pyramid of remedies set down in Chapter III, the following policy recommendations are proposed:

<u>Starting from restoration</u>

1. Restoration should be required whenever it is cost effective and technically feasible, but the criteria for cost effectiveness must be very clear in the law.

2. Environmental laws should provide incentives for optimal clean-up decisions by imposing liability for the long-term environmental costs of clean-up activities, especially in vulnerable ecosystems such as rocky shores and beaches where more scientific information is available. However, where the environmental damage caused by clean-up operations is likely to be irreversible (given the specific characteristics of the environment), a criminal sanction could be added for optimal deterrence (see below).

[4] 'Restoration has been elevated as the crisis response best practice in marine and coastal ecosystems'. See R.L. Wallace, S. Gilbert & J.E. Reynolds, 'Improving the Integration of Restoration and Conservation in Marine and Coastal Ecosystems: Lessons from the *Deepwater Horizon* Disaster', 69(11) *BioScience* 920 (2019). Arguably, this ecological scholarship draws on the lessons of Aldo Leopold whose final chapter "The Land Ethic" of A Sand County Almanac in 1949 called for a combined socio-ecological ethic of conservation and restoration to combat the degradation of ecosystems.

3. Environmental liability procedures should be designed to ensure that decisions on compensation for restoration costs are taken within a reasonable time. Moreover, such decisions should not depend on proof that the environment has recovered (either naturally or through human intervention).

4. Restoration obligations should be accompanied by enforcement mechanisms such as penalty payments for each day of delay. Of course, this will require the cooperation of public authorities for enforcement and monitoring.

5. Restoration should be incentivised over the long term and until full restoration is achieved, with the help of local people. Local people could be involved in restoration programmes based on a participatory process (consensus-based restoration). This in turn would reduce the need for monitoring by public authorities (point 4).

6. Informed parties (polluters and public administrations) must be properly incentivised to take immediate action from the first moment they become aware of the accident. Response actions must be clearly defined in advance at all administrative levels and late responses by public authorities should be penalised.

7. Competent public authorities must provide clear guidance to experts asked to carry out equivalence analyses for compensatory restoration.[5]

8. Competent authorities need to collect more and updated ecological data on the baseline conditions of ecosystems with critical thresholds that may suffer irreversible damage. This data must be made publicly available to local communities and potential polluters.

<u>Damages</u>

1. The obligation to pay monetary compensation (damages) should be imposed whenever restoration is not cost-effective/impossible, resulting in permanent losses. The calculation should be based on the Ecosystem Services Valuation Database (ESVD),

[5] The accuracy of the analysis and data collection in habitat equivalency should be determined by the scale of the injury. For instance, metrics should be chosen in such a way that the cost to achieve precision is equal to the complexity of the affected services, especially when it is difficult to find habitats with comparable quality, quantity and value of services.

which provides up-to-date and easily accessible estimates of the value of ecosystem services according to the type of damaged ecosystem (forest, wetland, etc.).

2. The obligation to pay damages should also cover interim losses from the time of the incident until full restoration. Interim losses can be calculated "on average" by multiplying the value of the damaged ecosystem by the number of years needed for full restoration.

3. Finally, the obligation to pay damages should be imposed whenever the application of equivalence analyses to the scale of restoration is likely to lead to gross errors due to the complexity of the ecosystem to be replaced by an alternative site.

Financial guarantees

Financial guarantees should cover both financially quantifiable environmental costs (clean-up and removal costs) and non-financially quantifiable environmental costs, such as interim losses and non-use values. The calculation of the latter can be based on the ESVD, especially for wetlands and forests. Financial guarantees must be mandatory, such as compulsory liability insurance.

Regulations/criminal sanctions

1. Environmental regulation remains crucial, as the probability of detecting liable parties may be less than one and liability laws may not provide optimal deterrence.

2. Non-monetary sanctions (criminal law) should be used to optimally deter irreversible environmental damage to ecosystems with unique biodiversity value and irreplaceable characteristics for which there may be no equivalent restoration or monetary compensation.

Final point for better restoration in the EU

In the EU, scholars are calling for more guidance and research on restoration.[6] Restoration involves a high degree of scientific judgement in choosing metrics, identifying baselines, targets and equivalencies for substitution. In addition, compensatory restoration 'has a relatively short history of only about three decades'[7] and many scientists, ecologists and economists, including those working in public agencies, are relatively unfamiliar with it.[8] For this reason, public authorities should synthesise the existing body of work and make it available to restoration practicioners.

Also, more monitoring of newly restored sites is needed to improve current restoration practices. Since the ELD has been scarcely applied and it is not possible to keep a record of historical incidents (as in the US), some scholars suggested that 'a series of scenarios should be generated to prepare rapid responses to any type of foreseeable incident'.[9]

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[6] See J. Lipton, E. Özdemiroğlu Ece, D. Chapman & J. Peers (eds), *Equivalency Methods For Environmental Liability : Assessing Damage And Compensation Under The European Environmental Liability Directive* (2019); K. Van Biervliet, D. Le Roy & P.A.L.D. Nunes, 'A Contingent Valuation Study on Accidental Oil Spill Along the Belgian Coast', in F. Maes (ed), *Marine Resource Damage Assesssment Liability and Compensation for Environmental Damages* (2005).

[7] E.P. English, C.H. Peterson & C.M. Voss, 'Ecology and Economics of Compensatory Restoration', NOAA Coastal Response Research Center, CRRC (2009), at 160.

[8] HEA is the most common tool for restoration scaling, but stated-preference methods have been examined far more thoroughly in the literature. See A. Bas, P. Gastineau, J. Hay & H. Levrel, 'Méthodes d'Équivalence et Compensation du Dommage Environnemental, 123 *Revue d'Économie Politique* 127, at 154 (2013).

[9] H. Aiking, E.H.P. Brans & E. Ozdemiroglu, 'Industrial Risk and Natural Resources: The EU Environmental Liability Directive As a Watershed?', 1 *Environmental Liability* 3, at 10 (2010). These conclusions are based on the analysis of the Doñana case study.

Limitations and avenues for future research

Finally, some important limitations of this research need to be mentioned. They represent the limits of this dissertation and the starting point for future research.

1. Completeness

The case analysis carried out in Part II was limited to a few selected laws and cases. Indeed, the primary objective of this dissertation was to outline a multi-level analysis, develop a theoretical model of remedies and possibly identify further avenues for more focused research, rather than to provide a comprehensive empirical analysis of existing remedies for environmental damage. As a result, the findings of this dissertation indicate that there is a significant gap in our knowledge of how judges compensate for environmental damage at the national level. In addition, it is not certain whether restoration is successful from an ecological point of view and whether compliance is followed up after judicial decisions on environmental liability. This information is currently not available, either to administrative authorities or to polluters. Therefore, this dissertation has highlighted the crucial need for a complete empirical analysis of case law at the international and national levels, with the aim of developing a dataset and showing trends in the judicial assessment of environmental damage and the follow-up of compliance.

2. Rationality and behavioural biases

Second, the positive analysis proposed was based on the assumption that polluters would respond rationally to the incentives offered by a smart combination of remedies. However, it would be interesting to examine whether possible behavioural biases might affect how polluters prevent environmental damage and respond to the different remedies in the pyramid. Similarly, it was assumed that public authorities would act in the public interest when carrying out clean-up operations and bringing actions for restoration or compensation. However, as mentioned above, it is possible that they could be captured by private interests, which would require a refinement of the proposed remedies.

3. Validity

Unfortunately, this research suffered from a major limitation in terms of information on the behaviour of polluters, either ex ante or ex post. While it has been argued that better incentives are needed to promote optimal deterrence and cost-effective remediation, there is a complete lack of information on whether liability lawsuits have the potential to induce such changes in the behaviours of polluters. This is an interesting point for future research, also with a view to providing policy makers with evidence on the validity of the pyramid of remedies for environmental damage proposed in this paper.

4. Judicial decision-making

Thirdly, this paper has not explored the reasons why tribunals have chosen certain valuation methods over others. For example, it remains unclear why the International Court of Justice did not appoint an independent expert in the Costa Rica/Nicaragua case and why courts tend to reject novel valuation methods (although some hypotheses can be made). This limitation clearly points to the need for further research into judges' personal motives in dealing with environmental compensation, their behavioural biases and their competence in such technical matters. It will then be possible to determine what changes are needed to make courts more efficient in dealing with environmental compensation.

5. Methodology of environmental damage assessment

Finally, this research did not provide a final solution on how environmental damage should be calculated by courts. Moreover, it did not consider several methods of valuing nature proposed by disciplines other than environmental and ecological economics (such as behavioural economics and psychology). While these alternative methods could have been investigated, the focus was only on methods that have been already used by courts or mentioned by the laws (such as the ecosystem services approach). In any case, I humbly believe that a multidisciplinary approach involving ecologists, psychologists, economists and lawyers would be required to answer the question. The way to an efficient, ecological and legal solution for the valuation of environmental damage is still open.

Final take-aways

The occurrence of environmental accidents raises two peculiar needs.
The first is to return the damaged environment back to the conditions prior to the accident.
The second is to provide polluters with optimal care and activity incentives.
These objectives have in common the minimisation of the total social costs of accidents, including the costs for the environment.
By examining existing liability laws and selected cases, this dissertation found out that the first objective is currently prioritised over the other one under international, US and EU liability laws.
However, a too strong focus on restoration may result in the other goal of environmental remedies (cost internalisation) being overlooked, with a risk of underdeterrence.
At the same time, environmental restoration does not seem to be achieved in a costeffective manner, meaning that polluters are not exposed to the full costs of restoration.
Eventually, a smart combination of remedies for environmental harm would help to achieve the two goals above by ensuring the full internalisation of the total economic value of nature.

List of Abbreviations

BP British Petroleum

CBD Convention on Biological Diversity

CERCLA Comprehensive Environmental Response, Compensation and Liability Act

CLC International Convention on Civil Liability for Oil Pollution Damage

CV Contingent Valuation

CWA Clean Water Act

DOI Department of Interior (US)

DPA Deepwater Port Act

DWA Deepwater Horizon

ECHR European Court of Human Rights

ECJ European Court of Justice

ELD EU Directive on Environmental Liability

EPA Environmental Protection Agency (US)

ESVD Ecosystem Service Valuation Database

EVOS Exxon Valdez Oil Spill

GHG Greenhouse Gas

HD EU Habitats Directive

HEA Habitat Equivalency Analysis

HP Hedonic Pricing

IACrtHR Inter-American Court of Human Rights

IAS Invasive Alien Species

ILD International Law Commission

ICJ International Court of Justice

ICSID International Centre for Settlement of Investment Dispute

INRA Institut National de la Recherche Agronomique

IOPCF International Oil Pollution Compensation Fund

IPBES Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services

ITLOS International Tribunal for the Law of the Sea

ITOPF International Tanker Owners Pollution Federation Limited

NOAA National Oceanic and Atmospheric Administration (US)

NRDA Natural Resource Damage Assessment

MoFF Indonesian Ministry of Environment and Forestry

MPF Brazilian Ministério Público Federal

MPRSA Marine Protection, Research and Sanctuaries Act

OECD Organisation for Economic Co-operation and Development

OPA Oil Pollution Act

PNCP French Port-Cros National Park

REA Resource Equivalency Analysis

REMEDE Resource Equivalency Methods in Environmental Damage Assessment

SCC Social Cost of Carbon

TAPAA Trans-Alaska Pipeline Authorization Act

TEEB The Economics of Ecosystems and Biodiversity

TEV Total Economic Value of nature

TCM Travel Cost Method

WBD EU Wild Birds Directive

WTA Willingness To Accept

WTP Willingness To Pay

UNCC United Nations Compensation Commission

UNCLOS United Nations Convention on the Law of the Sea

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Summary

Traditionally, the environmental damage has been addressed by regulations, administrative and criminal sanctions. However, these tools have been found to be inadequate for the purpose of remedying and preventing harm to natural resources. As a result, environmental liability provisions have been increasingly introduced to address pollution caused by events such as oil spills and toxic leakages. A first innovation has been to grant public bodies with legal standing to file claims for compensation of damage to public natural resources. Another significant improvement in the field of environmental liability has been the extension of the notion of compensable environmental damage to include non-use or passive-use values in the final amount of compensation. This has been accompanied by a parallel development of environmental economic scholarship, which has informed not only ex ante benefit-cost analyses for policies and projects, but also the ex-post valuation of environmental accidents. A significant challenge in this field since the 1970s has been the assessment of non-use or passive-use values, where traditional valuation methods based on observable behaviour are not applicable. New techniques to value the damage beyond market-based losses have been proposed and gradually refined. Economists have been debating the accuracy and reliability of these techniques for the past two decades. Concurrently, ecologists have been developing their own methods of valuation. In addition, an emerging tendency in the law is represented by the restoration-based compensation of environmental damage that appears to circumvent the contentious and time-consuming use of methods designed for non-use values (e.g., stated preference). Nevertheless, the exchange of information among these interconnected domains of knowledge is not as fluid and expeditious as it could be. Consequently, this research aims to provide an overview of the existing discrepancy between liability laws and economic scholarship. The specific research question is whether remedies for environmental damage at the international, regional and national levels are providing polluters with optimal care incentives to minimise the environmental costs of accidents while, at the same time, ensuring cost-effective restoration. Although limited, the resulting picture is quite diverse and comprises both positive and negative aspects. While some best practices have emerged at some levels of the law and in some countries, many challenges remain. They mainly relate to the notion of compensable environmental damage, the role of the economic valuation in the law and judicial practice, the multiple levels and branches of law intertwined in a single polluting event, and the private interests of all parties involved in the environmental damage assessment. Despite the complexity of the aforementioned issues, this research puts forward a novel theory of remedies for environmental damage that wishes to provide a smart solution to attain more efficient deterrence and adequate remediation.

Samenvating

Traditioneel is milieuschade aangepakt door middel van reguleringen, administratieve en strafsancties. Deze instrumenten bleken echter onvoldoende voor het herstel van en het voorkomen van schade aan natuurlijke hulpbronnen. Als gevolg hiervan zijn er steeds meer bepalingen voor milieu-aansprakelijkheid ingevoerd om vervuiling, veroorzaakt door gebeurtenissen zoals olielekkages en giftige lozingen, aan te pakken. Een eerste vernieuwing was het verlenen van rechtsbevoegdheid aan overheidsinstanties om vorderingen tot vergoeding van schade aan publieke natuurlijke rijkdommen in te dienen. Een andere belangrijke verbetering op het gebied van milieu-aansprakelijkheid was de uitbreiding van het begrip compensabele milieuschade om waarden van niet-gebruik of passief gebruik op te nemen in het uiteindelijke compensatiebedrag. Dit ging gepaard met een parallelle ontwikkeling van de milieu-economische wetenschap, die niet alleen ex-ante kosten-batenanalyses voor beleid en projecten heeft geïnformeerd, maar ook de ex-post waardering van milieurampen. Een significante uitdaging in dit veld sinds de jaren 1970 is de beoordeling van waarden voor niet-gebruik of passief gebruik waarbij traditionele waarderingsmethoden gebaseerd op waarneembaar gedrag niet toepasbaar zijn. Nieuwe technieken om schade te waarderen die verder gaan dan marktgebaseerde verliezen zijn voorgesteld en geleidelijk verfijnd. Economen hebben gedurende de afgelopen twee decennia gedebatteerd over de nauwkeurigheid en betrouwbaarheid van deze technieken. Tegelijkertijd hebben ecologen hun eigen waarderingsmethoden ontwikkeld. Een opkomende tendens in het recht is echter de op herstel gebaseerde compensatie van milieuschade, die het controversiële en meer tijdrovende gebruik van technieken die ontworpen zijn voor niet-gebruikswaarden (bijv. aangegeven voorkeur) lijkt te vermijden. Desalniettemin is de uitwisseling van informatie tussen al deze onderling verbonden kennisdomeinen niet zo vloeiend en snel als het zou kunnen zijn. Daarom streeft dit onderzoek ernaar een overzicht te bieden van de bestaande discrepantie tussen aansprakelijkheidswetten en economische wetenschap. De specifieke onderzoeksvraag is of remedies voor milieuschade op internationaal, regionaal en nationaal niveau vervuilers voorzien van optimale zorgprikkels om de milieukosten van incidenten te minimaliseren en tegelijkertijd kosteneffectief herstel te garanderen. Hoewel beperkt, is het resulterende beeld heel divers en bevat het zowel positieve als negatieve aspecten. Hoewel er op sommige niveaus van de wetgeving en in sommige landen *best practices* naar voren zijn gekomen, blijven er vele uitdagingen bestaan. Deze zijn voornamelijk gerelateerd aan de notie van compensabele milieuschade, de rol van de economische waardering in het recht en de rechtspraktijk, de verschillende niveaus en takken van het recht die verweven zijn in een enkele vervuilende gebeurtenis en de private belangen van alle partijen die betrokken zijn bij de beoordeling van milieuschade. Ondanks de complexiteit van de voornoemde kwesties, stelt dit onderzoek een nieuwe theorie van remedies voor

milieuschade voor die tot doel heeft een slimme oplossing te bieden om efficiëntere afschrikking en adequaat herstel te bereiken.

Appendix

Doctoral Activities 2018/2023

COURSES		
name course	*venue of the course*	*time period course*
Introductory Statistics	Bologna University	October 2018
Introduction to European Competition Law	Bologna University	October 2018
Environmental Economics	Bologna University	October – November 2018
Experimental Economics – Topics	Bologna University	November – December 2018
Modelling Private Law	Bologna University	November – December 2018
Global change of human-modified ecosystems	Bologna University – Department of Biology	November – December 2018
Ecology and Ecosystem Services	Bologna University – Department of Biology	November 2018
Game Theory, Behaviour and the Law	Bologna University	January 2019
Behavioural Law and Economics – Enforcement Mechanisms	Bologna University	January 2019
Law and Economic Development	Bologna University	February 2019
Empirical Legal Studies	Hamburg University	May 2019
Scientific Methods of Sustainable Decision-making	Hamburg University	June 2019
Statistics: fiction and facts	Hamburg University	June 2019
New Trends in Experimental Economics: survey experiments	Hamburg University	June 2019
Summer School of Law and Economics: - Teaching and research in L&E - Behavioural L&E - Empirical L&E	Hamburg University	July 2019 (4 weeks)

- Economics of relilgion		
ELFA Summer School on: "Critical topics in environmental law in comparative perspective"	Como	July 2019 (1 week)
ALTER-Net Summer School "Biodiversity and ecosystem services: science and its impact on policy and society"	Peyresq	August 2019 (ten days – 63 hours)
Research Data Management and Academic Integrity	Online course (Canvas)	September 2019
Academic Writing	Rotterdam University	September – November 2019
Managing your PhD	Rotterdam University	October – November 2019
Private L&E	Rotterdam University	University November 2019
Public L&E	Rotterdam University	November 2019
Advanced Data Analysis	Rotterdam University	December 2019
Course on Advanced Data Analysis	Rotterdam University	12-13 December 2019
Ius Commune Training (Foundations of Ius Commune)	Maastricht University	27 – 29 January 2020
Course on Advanced Empirical Methods Research Design	Rotterdam University	13 – 20 March 2020
EGSL online course on Communicate your Research	Online	April – June 2020
EGSL online course on Research Design	Online	April – August 2020
Ius Commune Training (Comparative Law Methodology)	Online (Utrecht University)	May 2020
Ius Commune Amsterdam Masterclass	Online (Amsterdam University)	26 June 2020
Environmental and resource economics (Campiglio e Xepapadeas)	Online (Bologna University)	September – November 2020
Resource valuation and decision-making methods (Montini – Xepapadeas)	Online (Bologna University)	September – November 2020

Research design (EGSL course)	Online (Rotterdam University)	September 2020 – January 2021
Workshop “Write a research grant”	Online (Rotterdam University)	17 September 2020
Workshop “How to use ORCID”	Online (Rotterdam University)	27 October 2020
Teaching Lab	Online (Rotterdam University)	November 2020 (4 classes)
Horizon Europe	Online (Bologna University)	End-November 2020 (1-week meetings)
Computational methods for lawyers (J. Kantorowicz)	Online (Rotterdam University)	14-16 December 2020
Oxford Winter School on Climate Change	Online (Oxford University)	January 2021 – March 2021 (8 weeks)
Machine Learning (Prof. Engel)	Online (Rotterdam University)	3-4-5 February 2021
Advanced Research Methods: advanced introduction to ELR (Prof. Klick)	Online (Rotterdam University)	March 2021 (4 classes)
Workshop Research Ethical Review	Online (Rotterdam University)	19 April 2021
Comparative Law Course	Online (EGSL – Rotterdam University)	29 April 2021
PhD Course on Economic Analysis of Corporate Misconduct with Prof. Arlen	Online (Norwegian School of Economics)	24-28 May 2021
Online Workshop on the Computational Analysis of Law (OWCAL)	Online (University of Virginia Law School)	13 May 2021
Course on “Accounting for Ecosystem Services: theoretical basis and practical applications in Europe”	Online (with the Joint Research Center of the EU)	6 June 2021

iCourts/PluriCourts PhD Summer School 2021	Online (Copenaghen University)	14-18 June 2021
Gerzensee Summer School on Law and Economics of Liability and Enforcement (by Prof. Arlen)	Online	21-25 June 2021
ATLAS Summer School (Empirical Legal Methodology)	Online (Rotterdam University)	24 – 30 June 2021
Summer Course on AI and Law	Online (EUI-Florence)	8 – 18 July 2021
UNEP course on restoration	Online (self-paced)	September – October 2021
Work management	Online (Rotterdam University)	October – November 2021
Training in “Evaluating Ecosystem Services with Remote Sensing”	Online (ARSET)	23, 25, 30 August 2022
‘Prioritizing restoration areas using Spatial Multicriteria Analysis”	Alicante	5 September 2022

CONFERENCES, SEMINARS AND OTHER ACTIVITIES (presentations)		
Description	*Venue*	*Date*
EDLE 3rd-year seminar	Bologna University	10 November 2018
ACE conference	Bologna University	15 – 16 November 2018
9th MetaLawEcon Workshop (presentation paper)	EUI - Florence	12 – 13 December 2018
SIDE – ISLE 14th annual conference	Lecce University	13 – 15 December 2018
Lectures of L&E at the Insitute of L&E	Hamburg University	May – July 2019
BACT seminar on "YouTube vs. Netflix: an empirical analysis of consumer behaviour"	Rotterdam University	19 September 2019
IPCC Special Report "Climate Change and Land"	Webinar CMCC	24 September 2019
REI conference "Shifting from Welfare to Social Investment States: the Privatization of Work-Related Risk control and its Impact on Inclusion"	Rotterdam University	26 – 27 September 2019
"The Democratic Courthouse? Unravelling the complex relationship between design, due process and dignity in English Courts Seminar with Linda Mulcahy"	Rotterdam University	27 September 2019
Guest Lecture "How informative is the text of securities complaints?" by Adam Badawi	Rotterdam University	3 October 2019
'The Quest for Controlled Freedom'	The Hague	11 October 2019
Ecosystem Services Partnership 10th World Conference on Ecosystem Services (presentation paper)	Hannover	20-25 October 2019

RDM (Research Data Management) Workshop	Rotterdam University	29 October 2019
Under Pressure – Lecture on Stress Management	Rotterdam University	29 October 2019
1st EDLE Seminar + BACT Seminar on 'More than the Money: Payoff-Irrelevant Terms in Relational Contracts' by Monika Leszczynska	Rotterdam University	31 October 2019
Conference on 'Courts and Government: The role of civil, criminal and (European) administrative law in redressing an alleged lack of good government'	Utrecht University	8 November 2019
24th Ius Commune Conference	Leuven University	28 – 29 November 2019
Lunch Lecture *Turning Gold into Green: The Responsibility of European Financial Supervision in Green Finance*	Rotterdam University	4 December 2019
2nd-year EDLE Seminar (presentation chapter)	Rotterdam University	5 December 2019
International Conference "*Private Law and Market Regulation in the face of contemporary grand challenges*"	Groningen University	9-10 December 2019
Workshop 'On the Crossroads of Law and Economics'	Rotterdam University	11 December 2019
BACT Seminar "Primary motives behind outsourcing legal rules"	Rotterdam University	12 December 2019
Workshop New RefWorks	Rotterdam University	17 December 2019
SIDE - ISLE 2019 - 15TH ANNUAL CONFERENCE (presentation paper)	Milano University (Statale)	19 – 21 December 2019

International Principles & Standards for the Practice of Ecological Restoration	Online (SER Europe)	21 January 2020
Forests for Biodiversity and Climate Change	European Parliament - Bruxelles	4-5 February 2020
Early Career Research Workshop (presentation paper)	Bruxelles (ULB)	13 – 14 February 2020
Poster Presentation	Rotterdam University	3 March 2020
2nd-year EDLE seminar (presentation chapter)	Rotterdam University	5 March 2020
Webinar with R. Hoekstra (founder of Metricsforthefuture)	Online (Rethinking Economics Rotterdam)	7 May 2020
Webinar Environmental Law Institute "Wetlands and Disaster Resilience"	Online (US Environmental Law Institute)	19 May 2020
Webinar Environment Europe Lecture Intellectual Roots of Ecological Economics	Online	21 May 2020
Webinar FAO on Soil Biodiversity	Online	22 May 2020
EDLE Seminar (presentation chapter)	Online	3 June 2020
Conference: Private Rights for Nature	Online (ACT Amsterdam Centre of Transformative Private Law)	3-5 June 2020
Webinar Valuing Nature Demystifying interdisciplinary working	Online	10 June 2020
BACT Seminar Rachlinski	Online (Rotterdam University)	25 June 2020

Ius Commune Amsterdam Masterclass (presentation)	Online (Amsterdam University)	26 June 2020
RILE Workshop On the Crossroad between Law and Economics	Online (Rotterdam University)	30 June 2020
Environmental policy and management teleconference (interview with me and prof. Sbokos)	Online (Umass Boston)	1 July 2020
InVEST Virtual Workshop (Natural Capital Project)	Online	21 July 2020
EDLE Summer Seminar with G. Dominioni	Online (Rotterdam University)	31 July 2020
AlterNet Virtual Summer School (short talks)	Online	31 August – 4 September 2020
Hamburg Lectures in L&	Online (Hamburg University)	Until July 2020
EALE Annual Conference	Online	24 – 25 September 2020
EU Green Week	Online	19 – 22 October 2020
3rd-year EDLE Seminar (presentation chapter)	Online (Bologna University)	6 November 2020
UAS Conference – PhD Workshop II "Sustainable development research: an interdisciplinary chance or challenge? (presentation paper)	Online (Freie Universität Berlin)	12 – 19 November 2020
Annual Ius Commune Congress – Workshop on Liability and Insurance (presentation paper)	Online (Maastricht University)	26-27 November 2020

Seminar: "The Promises & Pitfalls of Taxing Carbon"	Online (Rotterdam University)	1 December 2020
Seminar: “Superfund at 40: the future of Superfund, has the mission expanded and will it be permanent?”	Online (US Environmental Law Institute)	8 December 2020
BACT Seminar (prof. Jeroen Luyten)	Online (Rotterdam University)	10 December 2020
Young Legal Research Conference on Governing Societal Challenges in Transformational Times (presentation paper)	Online (Hasselt University)	21 December 2020
Hamburg lectures of L&E	Online (Hamburg University)	October – December 2020
Research funding days	Online (Rotterdam University)	26-27-28 January 2021
High-level Conference on the Future of Europe	Online (College of Europe)	28-29 January 2021
High Conference with Prof. Dasgupta	Online (Royal Society)	2 February 2021
Empirical Legal Studies Research Day	Online (Utrecht University)	5 February 2021
EMTM Presentation	Online (Hamburg University)	11 February 2021
Annual Conference IAERE	Online	18 February 2021
Rethinking Environmental Economics	Online	8 March 2021
We Value Nature 10-day challenge	Online (Capitals Coalition)	10-21 March 2021

PES for coral reefs	Online (CFA – Conservation Finance Alliance)	11 March 2021
BACT seminar - Marnix Hebly - Compensation and redress for damage	Online (Rotterdam University)	18 March 2021
Intelligenza Artificiale e Banche dati per la giustizia italiana	Online (Accademia 360)	19 March 2021
Urban Forestry Days	Online (European Forest Institute)	23 – 24 March 2021
Joint Seminar - The Future of L&E (presentation)	Online (Maastricht University)	25-26 March 2021
Comparative Torts - Liability for ecological harm	Online (British Association of Comparative Law)	30 March 2021
Carbon Pricing on Trial: Unpacking the Supreme Court Decision	Online (Smart Prosperity institute Canada)	30 March 2021
19th Annual Conference on European Tort Law	Online (Institute for European Tort Law)	8-9 April 2021
Workshop: "Article 47 of the EU Charter and effective judicial protection - The Court of Justice's perspective	Online (Maastricht University)	15-16 April 2021
Michigan Junior Scholars Conference (presentation paper)	Online (Michigan University)	16 – 17 April 2021

Ius Commune Café	Online (Maastricht University)	19 April 2021
BEEG Meeting (Environmentalists)	Online (Bologna University)	30 April 2021
YESS (Young Scholars on Ecosystem Services) Workshop	Online	6 – 7 May 2021
First European Webinar "Economics of Biodiversity: the Dasgupta Review"	Online (Toulouse School of Economics)	7 May 2021
FIDE Seminar	Online (Leiden University)	12 May 2021
Ius Commune Café	Online (Maastricht University)	18 May 2021
The Economic Valuation of Coasts	Online (US Environmental Law Institute)	26 May 2021
12th Annual Meeting of the Society for Environmental Law and Economics (presentation paper)	Online (University of Notre Dame Law School)	28 May 2021
Leuven Masterclass (presentation paper)	Online (Leuven University)	27 – 28 May 2021
IUCN Environmental Week	Online	31 May 2021 – 6 June 2021
3rd ESP Europe Conference (presentation paper)	Online (Tartu University)	7 – 10 June 2021
Natolin Conference on the EU Green Deal (presentation paper co-authored)	Online (College of Europe in Natolin)	21 June 2021
IUCN AEL 2021 (presentation paper)	Online	1-4 July 2021

Conference on the role of science in climate litigation	Online (BIICL)	16 – 17 July 2021
Bournemouth Conference on Scientific Uncertainties in Env Law	Online	8 – 10 September 2021
EDLE Seminar (presentation chapter)	Online (Rotterdam University)	12 October 2021
Ius Commune Annual Conference (presentation paper)	Online (Maastricht University)	26 October 2021
Webinar China and the EU	Online (Maastricht University)	1 December 2021
SIDE Italian Annual Conference (presenting paper)	Trento	15 – 17 December 2021
First International Workshop for Environmental Law (presentation paper)	Online	21 January 2022
Conference on Transparency and Scientific Uncertainty	Online (Maastricht University)	27 – 28 January 2022
Climate Change Litigation in Europe: Comparative & Sectoral Perspectives and the Way Forward	Online (BIICL)	18 – 19 February 2022
Seminar 'A plasticised future? How Can the EU Tackle the Plastic Problem in its Entirety?'	Online (ERA)	16 February 2022
Seminar 'Forest Fires in Mediterranean Europe: Building Up to the Courts?'	Online (Bruxelles University)	21 February 2022
Individual meeting with Josh Teitelbaum	Bologna University	16 March 2022
EDLE first year workshop	Bologna University	18 March 2022
EDLE Joint Seminar	Online	28 – 29 March 2022

Climate Litigation (seminar with Anna Savaresi and Joana Setzer)	Online	30 March 2022
21st Annual Conference on European Tort Law	Online	21 – 22 April 2022
Climate change workshop (presentation paper)	Bologna University	10 May 2022
Yale European Studies Graduate Fellows Conference at the Yale MacMillan Center (presentation paper)	Online (Yale University)	13 – 14 May 2022
Ucall Conference on Corporate Liability (presentation paper)	Utrecht University	19 May 2022
Seminar: Using the Law to Save the Planet (presentation paper)	Online (Rotterdam University)	20 May 2022
Lecture on EU Environmental Law	Chioggia University – Department of Marine Biology	31 May 2022
Alternet Conference – Transformative Changes for Biodiversity and Health (participation and organisation)	Ghent	14 – 17 June 2022
19th Joint Seminar EALE – Geneva Association. Pandemics – Liability and Insurance (presentation paper)	Wien University	23 – 24 June 2022
GLEA2022 (German Law and Economics Association) presentation paper	Online	7 – 8 July 2022
Courts as an Arena for Societal Change (presentation paper)	Leiden University	8 – 9 July 2022
2022 IUCN Academy of Environmental Law Colloquium Re-Imagining Environmental Law (presentation paper co-authored)	Online	11 – 15 July 2022
First Max Planck Law Conference for Young European Scholars 2022 (presentation paper)	Luxembourg (Max Planck Institute)	14 – 15 July 2022

SERE 2022 (13th Conference of the Society of Ecological Restoration) help in the organization of the conference and field study on restoration	Alicante	5 – 9 September 2022
9th EELF 2022 (presentation paper co-authored with Prof. Faure)	Tarragona	20 – 23 September 2022
4th ESP Europe Conference, hosting session on "Novel Contributions to Ecosystem Service Research"	Online	10 – 14 October 2022
Environmental Law Lunches organized with Nicola Harvey	Online	Every first Tuesday of the month from February until June 2022
VISITING AT ICOURTS	Copenhagen	27 February – 30 April 2023
Book launch with Emilia Justyna Powell 'The Peaceful Resolution of Territorial and Maritime Disputes'	iCourts	8 March 2023
Lunch seminar with Gabrielė Chlevickaitė Witness Evidence And Legal Decision Making: Empirical And Normative Analyses Of International Criminal Justice	iCourts	15 March 2023
Lunch Seminar 'The Efficiency of Remedies for Environmental Harm' (presentation)	iCourts	22 March 2023
Lunch seminar with Raphael Oidtmann Fighting Impunity Through Intermediaries – The European Union, International Criminal Justice, and the Rule of Law in Times of War	iCourts	12 April 2023
Breakfast Briefing with Hjalte Osborn Frandsen Governance in a Time of Rapid Expansion, Privatization and Militarization of Human Presence in Outer Space: Contemporary Issues in International Space Law	iCourts	13 April 2023
Seminar with Teresa Violante – Employing weak judicial review to manage conflicts of authority	iCourts	13 April 2023

between constitutional Courts and the Court of Justice https://jura.ku.dk/icourts/calendar/2023/imagineicourts-seminar-with-teresa-violante/		
Roundtable with Guillame Larouche - International Courts (Trans)formations: The Role of European Lawyers in the 'Fabrique' of the International Criminal Court	iCourts	25 April 2023
FELLOWSHIP AT MPI	Luxembourg	1 May – 30 June 2023
On-site Referentenrunde on the implementation of the Directive (EU) 2020/1828 on collective redress	Luxembourg (Max Planck Institute)	10 May 2023
Ius Commune Annual Conference	Maastricht	11-12 May 2023
DILDR Department Meeting on Venezuela/Council with Olivier Baillet and Interest Group Update for the IG on International Economic Adjudication	Luxembourg (Max Planck Institute)	17 May 2023
DSL Session on Research Design and Methods with Prof Fernanda Nicola and Prof Jeff Miller	Luxembourg (University)	17 May 2023
Book Launch - Researching the ECJ: Methodological Shifts and Law's Embeddedness (ed.s M.R. Madsen, F. Nicola, A. Vauchez)	Luxembourg (University)	17 May 2023
Human Rights Insights Lunchtime Seminar with Dr Anne Goedert, Luxembourg's Ambassador-at-large for Human Rights	Luxembourg (University)	24 May 2023
CPLJ Lecture Series - New Trends in Procedural Law: the Comparative Approach "The Introduction and New Patterns of Precedent Systems in the Procedural Law of the Traditional Civil Law Countries: Possibilities and Innovations for the Stare Decisis" Prof. de Castro Mendes	Luxembourg (Max Planck Institute)	12 June 2023
The Economic Incentives for Biodiversity (participation and public presentation)	Vilm, Germany (Federal Agency	13 – 16 June 2023

	for Nature Conservation)	
Guest Forum	Luxembourg (Max Planck Institute)	22 June 2023

PUBLIC PRESENTATIONS		
Title	Venue	Date
“Integrating Ecosystem Services in Judicial Reasoning: the European Environmental Liability Legislation”	10th World Conference of the Ecosystem Services Partnership /Hannover Universität	21 October 2019
“Environmental Policy and Management – interview with Prof. Sbokos”	University of Massachusetts – School of Environment (online)	01 July 2020
“Liability for Ecological Damage: Looking for Efficiency in Valuing and Litigating Natural Resources Damages”	Hamburg Universität - EMLE (European Master of Law and Economics) Midterm Meeting	11 February 2021
“The Judicial Approach to Environmental Damage Assessment: Between Efficiency and Equity”	Maastricht University - 13th Joint Seminar “The Future of Law and Economics”	26 March 2021
“Unlocking the Potential of Environmental Liability in Transformational Times”	University of Michigan Law School – 7th Annual Juniors Scholar Conference (online)	16 – 17 April 2021
“Liability for Ecological Damage: Looking for Efficiency in Valuing and Litigating Natural Resources Damages”	12th Annual Meeting of the Society for Environmental Law and Economics (online)	28 May 2021
“Law over Troubled Water:Territorial Sea, Exclusive Economic Zone, High Sea”	Università di Padova – Department of Marine Biology (Chioggia, Italy)	3 June 2021

"Unlocking the Potential of Liability Laws through the Ecosystem Services or How to Make the Polluter Fully Liable for Environmental Losses"	Ecosystem Services Partnership – Europe Conference 2021 (online)	9 June 2021
"Ecological Damage and Liability: a Law and Economics Multilevel Perspective over the Efficiency of Remedies"	IUCN AEL 18th Annual Colloqium – Track "Environmental Damage and Liability" (online)	2 July 2021
"Is Restoration the Economically Efficient Remedy for Environmental Liability?"	Annual Conference of the Italian Society of Law and Economics (Trento, Italy)	15 – 17 December 2021
"Soft to be Strong. A legal and economic analysis of the use of bilateral soft law in the EU environmental external action"	The Supranational Democracy Dialogue – Università del Salento (Brindisi, Italy)	8 – 9 May 2022
"Is Clean Energy Really Clean?The Challenge of No Net Loss in the EU Energy Transition"	Yale European Studies Graduate Fellows Conference at the Yale MacMillan Center (online)	13 – 14 May 2022
"Climate change? L'addition s'il vous plait! A law and economics perspective on the calculation of damages"	Ucall Conference on Corporate Liability - Utrecht University (Utrecht, Netherlands)	19 May 2022
"Is environmental liability up for the challenge?"	Online seminar: Using the law to save the planet – Erasmus University Rotterdam	20 May 2022
"Climate Law"	Università di Padova – Department of Marine Biology (Chioggia, Italy)	31 May 2022
"Fifty Shades of Restoration in Italy: an Analysis of Legal Tools for Transformative Change"	Alternet Conference – Transformative Changes for Biodiversity and Health (Ghent, Belgium)	14 – 17 June 2022

"Shall We Insure Biodiversity?"	19th Joint Seminar EALE – Geneva Association. Pandemics – Liability and Insurance (Wien, Austria)	23 – 24 June 2022
"The Economics of Remedies"	GLEA2022 (German Law and Economics Association – Nancy University (Nancy, France)	7 – 8 July 2022
"Efficiency and Justice in Environmental Damage Calculation"	Courts as an Arena for Societal Change – Leiden University (Leiden, Netherlands)	8 – 9 July 2022
"Re-imagining the Governance of Ecological Restoration in Uncertain Times" (with Eleonora Ciscato)	2022 IUCN Academy of Environmental Law Colloquium Re-Imagining Environmental Law (online)	11 – 15 July 2022
"Ecological Damage and Liability in the EU – a Law and Economics Perspective on Remedies"	First Max Planck Law Conference for Young European Scholars 2022 (MPI, Luxembourg)	14 – 15 July 2022
"Fifty Shades of Restoration in Italy – Analysis of Legal Tools and Drivers for Transformative Change" (with Eleonora Ciscato)	13th Conference of the Society of Ecological Restoration (Alicante, Spain)	5 – 9 September 2022
"Rethinking Remedies for Environmental Harm" (joint paper with Michael Faure)	9th EELF (European Environmental Law Forum) Annual Conference 2022 (Tarragona, Spain)	21 – 23 September 2022
"Efficient Remedies for Environmental Harm"	7th AFED (Association Française d'Économie du Droit) Annual Conference (Montpellier, France)	13 – 14 October 2022

WORK EXPERIENCES		
Description	*Where*	*Time period*
Teaching assistant "Institutions of Public Law" (Prof. Silvia Nicodemo) 15 hours	Bologna University	September 2018 – June 2019
Teaching Assistant "Public Finance" (Prof. Franzoni) 20 hours	Bologna University	September 2020 – July 2021
Teaching assistant "Foundations of Law - Private Law" (Prof. Al Mureden) 15 hours	Bologna University	September 2020 – July 2021
Teaching Assistant "International Law" (Prof. Tanzi) 40 hours	Bologna University	September 2021 – July 2022
Teaching Assistant "Public Finance" (Prof. Franzoni) 15 hours	Bologna University	September 2021 – July 2022
EKLIPSE Call for Knowledge (upon request of the French Agency for Biodiversity) Member of the Expert Working Group on biodiversity protection in the mitigation hierarchy	Online (weekly meetings)	June 2021 - Ongoing
Ecosystem Services Partnership - Task Force on Guidelines and Tools for Integrated Ecosystem Services Assessment (led by Dolf De Groot and Evangelia Drakou) Member of the Support Team	Online (meetings every three months)	May 2022 - Ongoing
Teaching assistant "Law and Economics of Corporate Governance" (Prof. Franzoni) 20 hours	Bologna University	September 2022 – July 2023

Teaching Assistant “Economics of Regulation” (Prof. Fiorentini) 20 hours	Bologna University	September 2022 – July 2023
Teaching Assistant “Principles of Law” (Prof. Roversi Monaco) 20 hours	Bologna University	September 2022 – July 2023

PUBLICATIONS	
(expected) 10/24	'Climate Change? L'Addition, S'Il Vous Plaît! A Law And Economics Account On The Calculation Of Damages', in E. de Jong (ed), *Corporate Responsibility and Liability in relation to Climate Change.*
(expected) 07/24	'Environmental Damage', in A. Marciano & G.B. Ramello (eds), *Encyclopedia of Law and Economics.* DOI: 10.1007/978-1-4614-7883-6
05/24	'The relationship of humans and non-human nature reflected in the nitrogen cycle' (coauthored with B. West, M. Bauer, C. Chalkiadakis, N. Dendoncker, T. M. González-Martínez, A. Mascarenhas, B. Phillips, T. Ploumi, C. Rodriguez, M. Sutton, M. Vandewalle, C-L. Washbourne), Ecosystem and People
04/24	'Informalisation of the European Environmental External Action', (coauthored with F. Spera), Natolin NEST Series College of Europe. ISBN 978-83-63128-10-4
04/24	'Unveiling the Loophole of Compensatory Restoration After Damage in the EU', in E. Cocciolo, J. Jaria-Manzano, A. De la Varga Pastor, M. Marques-Banque, (eds), *Rethinking Environmental Law: Connectivity, Intersections and Conflicts in the Global Environmental Crisis.* ISBN: 9781839704475
07/23	'Soft to be Strong: The Use of Bilateral Soft Law in the EU Environmental External Action' (coauthored with F. Spera), European Papers. ISSN: 2038-0461
04/23	'Legal assessment of the Proposal for an EU Nature Restoration Law: Report by the Legal Working Group of the Society for Ecological Restoration Europe', (coauthored with A. Cliquet, A. Aragao, C-H. Born, F. Bouquelle, E. Ciscato, K. Decleer, H. Dotinga, F. Fleurke, V. Mauerhofer, M. Meertens, A. Mendes, B. Queffelec, M. Reese, H. Schoukens, A. Trouwborst, G. van Hoorick & J. Verschuuren), Tilburg University Research Portal.
03/23	'State of knowledge regarding how we can improve adherence to the Mitigation Hierarchy, with a particular focus on the avoid stage' Report of the Eklipse Expert Working Group on the Mitigation hierarchy upon a request of the Office Français de la Biodiversité - French Agency of Biodiversity (coauthored with S. Savilaakso, J. Storie, B. Caitana Da Silva, S.

	Campagne, D. Depellegrin, D. Geneletti, I. Kagkalou, D. Lacarac, S. Luque, A. Scott, L. Zahvoyska) DOI: 10.5281/zenodo.7780894
02/23	'Comparing The Efficiency Of Remedies For Environmental Harm: US V. EU', *Comparative Law Review – Special Issue 'Rescuing Comparative Law And Economics? Exploring Successes And Failures Of An Interdisciplinary Experiment'* 13/1 ISSN: 2038-8993
01/23	'Is Clean Energy really Clean? The Challenge of No Net Loss in the EU Energy Transition', *Yale European Graduate Fellows Conference Journal 2022.*
12/22	'Valuing Environmental Damages: Fundamental Issues And Methods', *Erasmus Law Review – Special Issue 'Using The Law To Save The Planet: Legal Options To Address Climate Change And Ecological Destruction'* 3/22 DOI: 10.5553/ELR.000234 - ISSN: 2210-2671
03/21	'The External Dimension of the EU Green Deal', Working Paper 2/2021, Osservatorio OSORIN, Italian Ministry of Foreign Affairs. ISBN 979-12-5976-085-2
11/19	'Claiming Damages for Climate Change. A Law and Economics Perspective' (funded by the Institute of the European Democrats)
10/18	'The Junker App: A New Practice for Waste Management' (coauthored with F. Nante), in *White Paper On Good Practices In The Fields Of Environment And Energy In The EU Member States*, Student Energy Group of the College of Europe 2016-2017.
04/13	'Da 'qualcosa' a 'qualcuno', da 'qualcuno' a 'qualcosa'. Percorsi esatti ed errati sul concetto di persona', in P. Buongiorno & S. Losse (eds), *Fontes Iuris. Atti del VI Jahrestreffen Junger Romanistinnen und Romanisten*, Edizioni Scientifiche Italiane, 2013. ISBN: 9788849525847

Propositions

Stellingen behorende bij het proefschrift van Francesca Leucci

1. The compensation and remediation of environmental damage should be subject to rigorous economic scrutiny, rather than being influenced by private interests and other contingent factors.

2. Environmental restoration is currently prioritised over deterrence under international and EU liability laws.

3. Current liability laws do not expose polluters to the full cost of environmental accidents, including the costs beyond clean-up and restoration.

4. Post-spill cleaning operations tend to prioritise the use-values to the non-use values of nature. Also, post-spill monitoring is undersupplied.

5. Standing and procedural rights may not be enough if the incentives to file a lawsuit are insufficient.

6. Monetising the environment is accepted and considered fair in North America as much as it is looked at with suspicion in Europe.

7. One might inquire as to the degree of certainty present in science and law. However, a more pertinent question is whether it is possible to achieve certainty in the courtroom when such a state is lacking both in science and in law.

8. Property rights are not enough, contracts are needed. Contracts are not enough, liability is needed. Liability is not enough, insurance is needed. Insurance is not enough, regulation is needed. Regulation is not enough, criminal law is needed.

9. Since liability shapes the incentives of relevant actors, it has both a legal and an economic objective.

10. The Law and Economics works as a lighthouse that can turn miserable hunters into accurate navigators by shedding a light on what is behind and what is in front.

11. Read to develop yourself, do a PhD to know yourself.
(proposition not to be defended)

Roma, Museo Nazionale Romano presso le Terme di Diocleziano (2018)