



ΕΘΝΙΚΟ ΜΕΤΣΟΒΙΟ ΠΟΛΥΤΕΧΝΕΙΟ

ΣΧΟΛΗ ΝΑΥΠΗΓΩΝ ΜΗΧΑΝΟΛΟΓΩΝ ΜΗΧΑΝΙΚΩΝ

**ΠΟΣΟΤΙΚΟ ΠΛΑΙΣΙΟ ΔΙΑΧΕΙΡΙΣΗΣ
ΝΑΥΤΙΚΟΥ ΡΙΣΚΟΥ ΓΙΑ ΤΗΝ
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*« Η έγκριση της διδακτορικής διατριβής από την Ανώτατη Σχολή Ναυπηγών
Μηχανολόγων Μηχανικών του Ε.Μ. Πολυτεχνείου δεν υποδηλώνει αποδοχή των
γνωμών του συγγραφέα (Ν. 5343/1932, Άρθρο 202) »*

ΠΕΡΙΛΗΨΗ

Η ναυτιλία, όπως κάθε άλλη ανθρώπινη δραστηριότητα, ενέχει κάποιο ρίσκο για την ανθρώπινη υγεία και το περιβάλλον. Οι καθημερινές ναυτιλιακές δραστηριότητες συνεπάγονται άμεσο ρίσκο, αν και στατιστικά μικρό, για την υγεία των εργαζομένων και των επιβατών. Το θέμα αυτό έχει αναλυθεί αρκετά στη σχετική βιβλιογραφία.

Επιπλέον, κατά τις τελευταίες δεκαετίες, υπάρχει μια αυξανόμενη ανησυχία σχετικά με τους κινδύνους των ανθρώπινων δραστηριοτήτων στο περιβάλλον, ιδίως της ρύπανσης από πετρελαιοκηλίδες και από τις αέριες εκπομπές των πλοίων. Για τις αέριες εκπομπές εξετάζονται κυρίως οι εκπομπές Διοξειδίου του Άνθρακα λόγω της συμβολής τους στο φαινόμενο του Θερμοκηπίου και στην Κλιματική Αλλαγή.

Τα άτομα και την κοινωνία στο σύνολό της οφείλουν να προσπαθήσουν να εφαρμόσουν μέτρα για τη μείωση σχετικών ρίσκων. Τα μέτρα αυτά θα πρέπει να είναι και αποτελεσματικά στη μείωση του κινδύνου αλλά και οικονομικά υγιείς επενδύσεις. Ως εκ τούτου, είναι αναγκαίο να αποτιμήσουμε αλλά και να αξιολογήσουμε εναλλακτικές λύσεις. Οικονομική αποτίμηση είναι η διαδικασία της εκτίμησης (συνήθως σε χρηματικούς όρους) του κόστους και του οφέλους των μέτρων. Το επόμενο βήμα είναι η οικονομική αξιολόγηση των εναλλακτικών μέτρων, προκειμένου να προταθούν για εφαρμογή όσα είναι οικονομικά βιώσιμα. Οι μέθοδοι που μπορούν να χρησιμοποιηθούν συνδέονται με τη ανάλυση του κόστους και της ωφέλειας (Cost Benefit Analysis, CBA) και με την αξιολόγηση της αποτελεσματικότητας (Cost Effectiveness Analysis, CEA).

Στο επίκεντρο λοιπόν της παρούσας διδακτορικής διατριβής βρίσκονται οι παραπάνω μέθοδοι και ο τρόπος εφαρμογής τους για την αποτίμηση του ρίσκου στην ανθρώπινη υγεία, και στο περιβάλλον μέσω της ρύπανσης από πετρελαιοκηλίδες και από αέριους ρύπους. Παρουσιάζεται μια κριτική στον τρόπο που αυτές χρησιμοποιούνται σήμερα και προτείνεται ένα πλαίσιο διαχείρισης του ρίσκου στη ναυτιλία με άξονα την οικονομική αποτίμηση και αξιολόγηση του ρίσκου. Αξιολόγηση συνεισφορά στη βιβλιογραφία είναι η έρευνα της μη γραμμική προσέγγιση της αποτίμησης του ρίσκου και η ανάδειξη των πλεονεκτημάτων της Ανάλυσης Κόστους Ωφέλειας (CBA) σε σχέση με την Ανάλυση Κόστους Αποτελεσματικότητας (CEA) που κατα κόρον χρησιμοποιείται σήμερα στη ναυτιλία

Ιδιαίτερο ενδιαφέρον έχει η έρευνα που σχετίζεται με την εκτίμηση των περιβαλλοντικών ζημιών από πετρελαιοκηλίδες η οποία μάλιστα έχει αποτελέσει τη βάση υποβολών της ελληνικής αντιπροσωπείας στον Διεθνή Ναυτιλιακό Οργανισμό και μέρος της οποίας περιλαμβάνεται στο τελικό κείμενο των τροποποιήσεων της μεθόδου Τυπικής Αποτίμησης του Ρίσκου (Formal Safety Assessment, FSA) που χρησιμοποιείται για την αποτίμηση κανονισμών.

Τέλος, παρουσιάζεται ένα γενικευμένο πλαίσιο καθώς και παραδείγματα για την αποτίμηση του ρίσκου και την αξιολόγηση των συναφών μέτρων. Το πλαίσιο αυτό υποστηρίζεται από την παρουσίαση των ισχυρών θεμέλιων του στην οικονομική θεωρία και πιο συγκεκριμένα στους τομείς της οικονομικής της ευημερίας και του περιβάλλοντος. Έτσι, το πλαίσιο που παρουσιάζεται θα συμβάλει στην λήψη τεκμηριωμένων αποφάσεων για την αντιμετώπιση του ρίσκου και το οποίο μπορεί εύκολα να εφαρμοστεί τόσο για το συμφέρον των μεμονωμένων επιχειρήσεων όσο και της κοινωνίας συνολικά.

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Quantitative Risk Management Framework for Maritime Safety and Environmental Protection

PhD Dissertation

PhD Dissertation. December 2011

National Technical University of Athens
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ABSTRACT

Shipping, as any other human activity, poses some risk to human health and the environment. Risk is everywhere and always has been as our modern society is a risk society (Beck, 1986). Daily maritime activities involve direct health risks to mariners and passengers; an issue that has been extensively addressed in the literature. During the last decades, there has also been an increasing concern on the harm of human activities to the environment, especially of waterborne pollution from accidental oil outflow and ship air emissions as a byproduct of burning bunker fuels.

Individuals and society as a whole do try to implement measures to reduce such risks. These measures should be both effective in reducing the risk and economically sound investments. Therefore it is necessary to 'value' and 'evaluate' alternatives that have the potential to reduce the relevant risk. Valuation is the process of estimating (usually in monetary terms) the costs and the benefits of risk reduction measures. The next step is the economic evaluation of the alternatives in order to propose the ones that should be implemented. The methods that can be used are associated with weighting the benefits of the control measures against the relevant costs (Cost Benefit Analysis, CBA) and assessing their efficiency (Cost-Effectiveness Analysis, CEA).

The focus of this work is to present the above methods within the maritime risk assessment frameworks. The current approaches are critically reviewed and a correct way to apply them is presented. For the first time, a non linear approach of valuation is investigated and the need to move towards CBA within risk assessment is justified. State-of-the art work performed in the context of the International Maritime Organization is presented, so as to address the issue of non linearities in the case of environmental damages from oil spills, along with some illustrative examples.

Finally, a generalized framework to address risk to human life and the environment is presented. This framework is supported by presenting its strong foundations in the economic theory and more precisely in the fields of welfare economics, public policy and environmental economics. Thus, we provide a solid way to support risk-informed decision-making to address the problems above which can be easily applied both for the interest of individuals and of society.

ACKNOWLEDGMENTS

I would like to express my deep appreciation to my mentor, Professor Harilaos N. Psaraftis, for his advice during my doctoral research endeavor for the past four years. As my supervisor, he has constantly forced me to remain focused on achieving my goal. There have been numerous occasions where I remember feeling disheartened and stumped about the direction of my research. Even a short discussion with him reinvigorated my enthusiasm. In addition, his observations, comments and constant guidance on my academic development make me feel incredibly privileged to have him as my supervisor. For what it's worth, and it is a lot to me, I dedicate this thesis to him.

I would also like to express my heartfelt appreciation to the other two members of my thesis advisory committee Professor Apostolos Papanikolaou and Assistant Professor Dimitrios Lyridis, whose comments and guidance have been instrumental in various ways.

I have also benefited greatly from the fruitful discussions with Associate Professor Nikolaos Ventikos. I am indebted for his guidance and support, also in personal matters.

The atmosphere within the Laboratory of Maritime Transport (LMT) at the Technical University of Athens (NTUA) has made my research very pleasant and I am grateful to all and everyone I worked with during my stay in there.

I have also benefited from discussions in various topics with Jorge Balleio, Jack Devanney, George Gratsos, Rainer Hamann, Pierre Sames, Peter Securius, Yasuhira Yamada and Panos Zachariadis.

In addition, I would like to thank my friends and family for their support and encouragement. Very special thanks to my girlfriend for the fruitful discussions but also patience and encouragement.

This work has been supported financially and in part by Det Norske Veritas. Their support is gratefully acknowledged.

Finally, note that parts of this dissertation are based on other published or unpublished work as shown below. Permission has been given by the co-authors to reproduce part of the relevant material in this dissertation.

Chapter 1 draws some material from:

1. Kontovas, C.A (2005) "Formal Safety Assessment : Critical Review and Future Role", Diploma Thesis, NTUA, June 2005 [unpublished].
2. Kontovas, C.A., Psaraftis, H.N. (2009) "Formal Safety Assessment: A critical review", *Marine Technology*, Vol. 46, No 1, January 2009.

Chapter 4 is based mainly on material from:

1. Kontovas, C.A., Psaraftis, H.N. , Ventikos N. (2010) "An Empirical Analysis of IOPCF Oil Spill Cost Data", *Marine Pollution Bulletin*, Volume 60, Issue 9, pp 1455-1466, September 2010.
2. "Further experience with non-linear oil spill cost functions", MEPC 61/18/2 (Dec. 18, 2009), Submitted to the IMO by Greece [unpublished]
3. Kontovas, C.A. and Psaraftis H.N. (2008), "Marine Environment Risk Assessment: A Survey on the Disutility Cost of Oil Spills", 2nd Int. Symposium on Ship Operations, Management and Economics, Organized by Society of Naval Architects and Marine Engineers, Athens, September 17-18, 2008.
4. Kontovas, C.A., Ventikos, N. , and Psaraftis H.N. (2011) ,"An Updated Analysis of IOPCF Oil Spill Data: Estimation of the Disutility Cost of Tanker Oil Spills ", IMAM 2011, International Maritime Association of the Mediterranean, Genoa, Italy, September 13-16, 2011.
5. Kontovas, C.A., Ventikos, N. , and Psaraftis H.N. (2010) ,"Estimating the Consequence Costs of Oil Spills from Tankers", 2011 SNAME Annual Meeting & Expo, Nov. 18-19, 2011, Houston, TX.

Chapter 5 draws mainly from :

1. Kontovas, C.A., Psaraftis, H.N., "Carbon Dioxide Emissions Valuation and its Uses", The 3rd International Symposium on Ship Operations, Management & Economics, Athens, Greece 7-8 October, 2010.
2. Kontovas, C.A and Psaraftis, H.N. (2011), "Climate Change Policy in Shipping focusing on emission standards and technology measures", *Environmental Engineering and Management Journal*, Vol. 10, No. 10, pp 1589-1596, October 2011.
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Athens, December 2011

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1

INTRODUCTION

1.1 Objectives and contribution of this dissertation

The management of safety at sea and the protection of marine environment are based on a set of accepted rules that are, in general, agreed through the International Maritime Organization (IMO), which is a United Nations organization established in 1948. Decisions on regulations are usually made in the Maritime Safety Committee (MSC) for matters concerning maritime safety and in the Marine Environment Protection Committee (MEPC) for those concerning the protection of the marine environment. The IMO has no enforcement authority, that being left to its member states, or to bodies like the European Union, that can adopt specific legislation and have the capability and legal authority to enforce compliance. While it is generally accepted that the overall level of maritime safety has improved in recent years, further improvements are still desirable. However, it can be argued that much of maritime safety policy worldwide has been developed in the aftermath of serious accidents (such as *'Exxon Valdez'*, *'Estonia'*, *'Erika'* and *'Prestige'*).

The need for proactivity has been argued extensively time and again (among others, see Psaraftis (2002) before *'Prestige'* and Psaraftis (2006) after *'Prestige'* for an analysis of the main issues). Even though it is generally wise to draw lessons from past accidents and attempt to use these lessons to improve regulations, many times industry circles have questioned the wisdom of such an approach. Why should the maritime industry and, in general, society, have to wait for an accident to occur in order to modify existing rules or propose new ones? The safety culture of anticipating hazards rather than waiting for accidents to reveal them has been widely used in other industries such as the nuclear and the aerospace industries and in many cases, within private firms. A proactive policy approach to risk is through effective risk management; with risk assessment being a major part of this procedure. Such an approach has also been implemented within the international shipping industry through what is known as *'Formal Safety Assessment'* (FSA).

There is a vast literature on risk analysis and assessment not only for the maritime industry but also for all risk-related industries. Key references on risk assessment are Bedford and Cooke (2001), and Aven and Vinnem (2007), Vose (2008), Cox (2006,2009) and Aven (2009;2011). Regarding maritime risk assessment the reader may refer to the risk assessment Guidelines published by class societies for example DNV/HSE(2001), ABS(2003) and ClassNK (2009) and to the excellent books of Kristiansen (2005) and Papanikolaou (2009).

In addition, a very well written and up to date literature review on risk assessment and the relative definitions can be found in Johansen (2010) and, on the issue maritime risk assessment, the reader is also referred to the review performed by the author in Kontovas (2005).

The critique on most of the relevant maritime risk techniques (including FSA) is strongly related to the methods used to estimate the cost effectiveness of the risk reduction measures, for a discussion of the limitations and deficiencies of FSA see Kontovas (2005), Kontovas and Psaraftis (2006, 2008, 2009), Giannakopoulos et al. (2007) and Zachariadis et al. (2009). Indeed, the most important step in a risk assessment is the one that leads to recommendations on which measures should be implemented. In this context, not only the methods used need improvement, but there is also a necessity to extend their focus beyond safety. The most important effects that will be looked at in the context of this dissertation are those related to environmental pollution.

The aim of this dissertation is to develop methods that can better assist the management of maritime risk, in order to protect both human life and the environment, with a focus on maritime oil spills and ship air emissions. For an effective protection, it is necessary to 'value' and 'evaluate' alternatives that have the potential to reduce the relevant risk. Valuation is the process of estimating (usually in monetary terms) the costs and the benefits of specific risk reduction measures. In all cases the residual risk should be below the acceptable thresholds (see Section 1.4). On the other hand, evaluation is to compare among alternatives and propose for implementation the best of them, according to some criteria. The basic methods that can be used are associated with weighting the benefits of the control measures with the relevant costs and benefits (Cost benefit Analysis, CBA) and assessing their efficiency (Cost-Effectiveness Analysis).

At a theoretical level, both valuation and evaluation are presented in a generic way so that they can be used with all major market and non-market goods. They are

presented together with the underlying economic theory so as to verify their solid foundations. Focus is not only on the individual firm or an individual but mainly society as a whole. Thus, the results can be used in economic appraisal of actions (projects) for private use and in regulatory decision-making. These are also viewed through the lens of uncertainty in the effects. Therefore, although the methods presented can be used in a standalone form, the illustrative examples are basically concerned with placing them within the framework of risk assessment. Given that the major tool used in the maritime industry is the so-called Formal Safety Assessment we focus on the way that the relative techniques can be used within FSA. There is no loss of generality as FSA is a typical risk assessment technique (as defined by the relevant ISO standard).

At a practical level, specific methodologies are developed in the context of regulatory decision making within the International Maritime Organization (IMO). The specific context here is the use of environmental risk evaluation criteria in Formal Safety Assessment (FSA) with a main focus on oil pollution, and, to a lesser extent, on ship air emissions. In fact, some of the work presented in here has been used by the IMO in recent decisions of the Marine Environment Protection Committee (MEPC) and more specifically in the draft amendment of the FSA Guidelines, which constitutes, in our opinion, a major contribution.

In more detail, ultimately, the aim of risk management within the scope of this work is the prevention of human fatalities and injuries, as regards human safety, and the protection of the environment by preventing oil spills and reducing ship air emissions, especially carbon dioxide emissions that contribute to the Climate Change. To achieve this, in the context of a huge number of relevant hazards and the inherited uncertainty, is not so straightforward. On top of that, the measures that could be implemented to reduce the risk posed by these hazards should both be effective in terms of reducing the risk and be economically sound investments.

To start with, we examine the use of non-linear valuation to estimate the benefit of the proposed actions and regulations as a result of the understanding that many damage functions are non-linear functions of the amount of harm; this is something rather new for maritime risk assessment. In addition, Illustrative examples are also presented in order to show that this can be easily applied in the traditional risk assessment.

Another major part of this work focuses on the two main evaluation techniques used, namely the Cost Effectiveness Analysis and Cost Benefit Analysis, performs a critical review of the current way of thinking within maritime risk assessment and highlights

the need to move towards the latter. The advocacy of an approach based on cost-benefit analysis in the regulation of environmental, health and safety contexts is nothing new (see Arrow, 1997). Although the literature review showed that a numerous studies have been conducted on maritime risk assessment, the literature in the relevant fields of valuation and evaluation of efficient policy measures is rather limited. In addition, traditional cost benefit analysis is not largely accepted by the maritime industry within risk assessment.

In effect, the above ties the practical way of conducting risk assessments with the strong theoretical background provided by economic theories (mainly from Welfare and Environmental Economics) and identifies possible ways to monetize and evaluate risk in such a practical way that can be easily incorporated in generic risk assessment frameworks. This work presents some state-of-the-art work on these issues that extends the traditional focus of maritime risk assessment on human health to environmental issues. Thus, contributes to a better understanding of the relevant risks and efficient risk-informed decision-making. Note that the work presented can be easily extended to cover many other environmental problems. In addition, the methods presented can be used both in a societal context (risk regulation) and within an individual firm.

The research presented also tries to elucidate some commonly used notions and propositions and to clarify the use of some major techniques that are unfortunately misused within maritime risk assessment. It may also seem that this dissertation advocates the use of Cost Benefit Analysis (CBA). This may be superficially true but the real aim is to advocate the use of the correct techniques and in the appropriate way of using both CBA and Cost Effectiveness Analysis (CEA); and it seems that maritime risk assessment has a real need for CBA.

To sum up, this work provides solid proposals on valuating and evaluating risk in a framework that may combine all major effects and helps in understanding the terminology and methods in such a way that may help to increase the acceptance of the related notions and techniques as an input to the decision-making process for environmental, health, and safety regulations. This will lead to more effective communication between risk assessors and between them, the decision-makers and the public and, thus, in better decisions.

The rest of this dissertation is structured as follows:

The rest of the **Introductory Chapter** presents some basic risk-related definitions, the relative risk assessment frameworks and Formal Safety Assessment, which is the major framework used in the maritime industry. In addition, we present the notion of the acceptable risk and the way that economic appraisal may be used within public policy and risk assessment.

Chapter 2 presents the two relevant valuation techniques, namely the Cost Benefit Analysis (CBA) and the Cost Effectiveness Analysis (CEA). Applications are drawn within the maritime risk assessment framework of FSA, in which these techniques have been used to assess the risk of human life.

Chapter 3 deals with the effects on human life and the very important issue of identifying cost effective measures to reduce risks that are posed to humans.

Chapter 4 discusses the effects of oil pollution on the environment and comments on the way of valuating the effects of oil spills based on the latest work within the IMO. Note that most of this work has been submitted to the IMO via Greece.

Chapter 5 deals with CO₂ emissions and the relevant Climate Change phenomenon. Ways to estimate and weight the benefits of measure that reduce carbon dioxide emissions are presented. Currently, there is no FSA study that deals with this subject although some cost-effectiveness criteria do appear in the literature.

Chapter 6 deals with both the theoretical and the practical way to incorporate the work presented in the previous Sections within maritime risk assessment (including FSA) and probabilistic oil risk analysis.

Finally, **Chapter 7** concludes this dissertation by presenting the conclusions and some possible areas for future research.

1.2 Some basic definitions

There is no doubt that the most important notion in risk assessment is that of 'risk'. According to the ISO terminology (ISO Guide 73:2009) the pivotal definition of risk is as the *"effect of uncertainty on objectives"*. Risk, as generally accepted, is *"often characterized by reference to potential events and consequences, or a combination of these"*, see Note 3 of ISO terminology. Clearly there are many definitions of risk and there has been a lot of discussion on proposed definitions and terms that are used within these definitions such as the concept of probability, frequency and uncertainty. The meaning of consequence is, on the other hand, quite straightforward. The author has presented these various definitions in Kontovas (2005).

Among the most commonly used ways of interpreting a probability, the two major ones are (Bedford and Cooke, 2001; Aven, 2003):

- (a) Probability is interpreted as a 'relative frequency' which is the relative fraction of times the event occur under the assumption that the situation that is being analyzed is hypothetically "repeated" for an infinite number of times.
- (b) Probability P is a measure of uncertainty about future events and consequences and is a subjective measure of uncertainty, conditional on the background knowledge of the risk assessor (that is related to the Bayesian perspective).

However, in many risk assessment frameworks (including FSA which is the major technique used in the maritime industry and will be presented in Section 1.3), the concept of 'frequency' seems prevalent, as risk is defined as *"the combination of the frequency and the severity of consequence"*, with frequency being defined in terms of accidents (rather than casualties). We note that this is not the standard definition of risk that appears in decision analysis, in which risk is defined as the combination of probability of occurrence and severity of consequence (see, for instance, Raiffa, 1968 and the discussion above). Furthermore, in maritime risk assessment 'risk' is normally presented *"as the product of the consequences and the probability of occurrence"* (Kristiansen, 2004). Kaplan and Garrick (1981) define risk as a "set of triplets", a set of scenarios S_i , each of which has a probability P_i and a consequence X_i . Furthermore, their approach is to use the frequency with which an event might take place which is essentially the notion of uncertainty about the frequency which is the 'probability of frequency'. Therefore, a risk analysis (which is a part of risk assessment) tries to answer the following questions (Kaplan and Garrick, 1981; Bedford, 2001): (i) What can happen, (ii) How likely is it to happen? and (iii) Given that it occurs, what are the consequences.

1.3 Generic Risk Assessment Frameworks

Most reviewers date the birth of risk assessment approximately in the early 1970s, when as a result of public and political pressures the US Atomic Energy Commission initiated a new era in safety of the nuclear power plants through a comprehensive study called 'Reactor Safety Study' (WASH-1400, or 'Rasmussen Study'—after its study leader Professor Norman Rasmussen of MIT), see Keller and Modarres (2005) and Apostolakis (2004). Since then, methodologies have advanced and the fields of application broadened into probabilistic risk assessment (PRA) of space systems and quantitative risk assessment (QRA) in the offshore oil and gas industry and assessments of human and environmental risk from chemicals, to mention a few (Kontovas, 2005).

Regulators (decision makers) and scientists advocate quantitative risk assessment (QRA) as providing both a logical framework and a systematic way for applying sound scientific knowledge and engineering experience to improve "rational" decision making among alternative decisions under uncertainty.

Most risk assessment frameworks share some common ground, which is conceptualized in Fig. 1-1 that presents risk assessment within the risk management process according to the latest ISO standard (ISO 31000:2009). In 2009, the new globally accepted ISO standard came together with a new associated vocabulary or relative terminology (ISO Guide 73:2009) and IEC 31010:2009, a supporting standard for ISO 31000 that provides guidance on selection and application of systematic techniques for risk assessment. ISO 31000 (2009) defines Risk Assessment (RA) as the "*overall process of risk identification, risk analysis and risk evaluation*" and is put within the wider context of Risk Management (RM) which the ISO Guide 73:2009 defines as the "*coordinated activities to direct and control an organization with regard to risk*".

So what is the reason for undertaking a risk assessment? Some researchers (e.g. Bley et al., 1992) state that the only reason is to understand a risk in order to do something about it and some other (e.g. Aven, 2010) disagree by saying that risk reduction is never a goal in itself and that the purpose of risk assessment is to provide input to a particular decision in a larger context. It seems that the latter position is confirmed by many organizations (including the IMO) stating that risk assessment aims at giving recommendations to relevant decision makers for safety improvements and can be used as a tool in (a) the evaluation of new regulations for maritime safety and protection of the marine environment and (b) comparing between existing and possibly improved regulations.

Risk assessment frameworks are, in general, very similar to each other and can be illustrated by the generic framework presented in Fig. 1-1. In the next Section, the main risk assessment tool that is being used in maritime industry –that of Formal Safety Assessment- will be presented. FSA has been extensively investigated by the author in Kontovas (2005) and Kontovas and Psaraftis (2009). The exact Steps of Risk Assessment will be analyzed in Section 1.3 when ‘Formal Safety Assessment’ will be introduced.

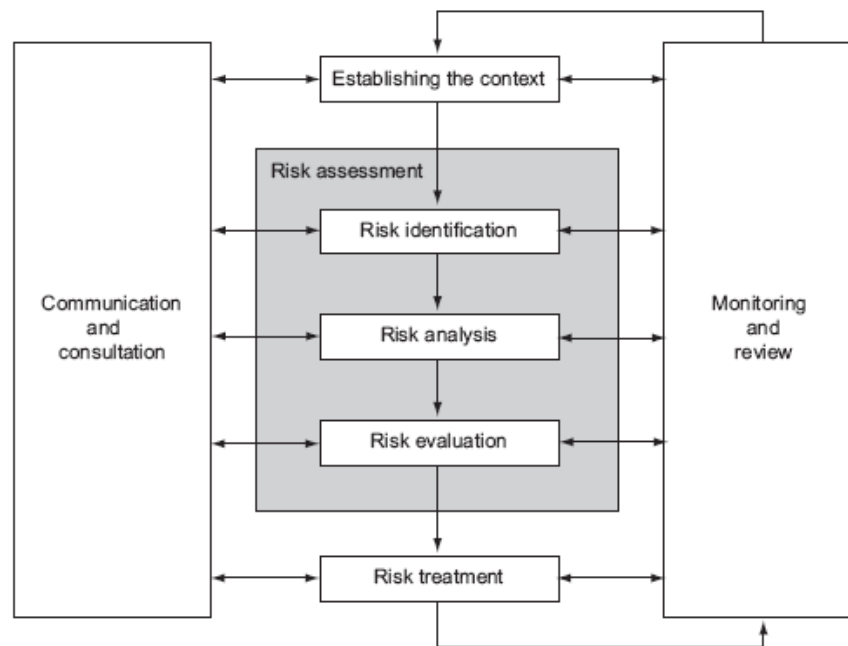


Fig. 1-1: Risk Management Process ISO 31000:2009 -Source: ISO (2009a)

Note that the work presented in this dissertation can be used within all relative frameworks that include valuation of the harm to human life and the environment (regarding oil spills and air emissions) and can be easily extended to other harms as well.

Finally, it is out of the scope of this dissertation to present in detail the Risk Assessment frameworks. There is a vast amount of literature on this area and a literature review on this issue has been also performed by the author in Kontovas (2005). The reader may be referred to Leitch (2010) and Purdy (2010) regarding the new international ISO standard on risk management. For some definitions of risk see Kaplan and Garrick(1981), Apostolakis (1990), Apostolakis and Wu (1993), Kaplan (1992),Adams (1995), Klinke & Renn (2002), Apostolakis (2004) and Haimes (2009). Finally, a very well written and up to date literature review on risk assessment and the relative definitions can be found in Johansen (2010). Key references on risk assessment are Bedford and Cooke (2001), and Aven and Vinnem (2007), Vose (2008) and Aven (2011).

1.4 Formal Safety Assessment

Formal Safety Assessment (FSA) was introduced by the International Maritime Organization (IMO) as “a rational and systematic process for accessing the risk related to maritime safety and the protection of the marine environment and for evaluating the costs and benefits of IMO’s options for reducing these risks”(see IMO, 2007). FSA aims at giving recommendations to relevant decision makers for safety improvements under the condition that the recommended measures (risk control options) reduce risk to the “desired level” and are cost-effective. FSA is, currently, the major risk assessment tool that is being used for policy-making within the IMO, however, until now its main focus was on assessing the safety of human life. No environmental considerations have been incorporated thus far into FSA guidelines. Also note that FSA exhibits some limitations and deficiencies. The reader is referred to Kontovas (2005) and Kontovas and Psaraftis (2006,2008 and 2009) for a discussion on these issues.

To achieve the above objectives, the IMO’s guidelines on the application of the FSA recommended a five-step approach (which is illustrated in the following Figure) and consists of:

1. Hazard Identification
2. Risk Assessment
3. Risk Control Options
4. Cost-benefit Assessment
5. Recommendations for decision making

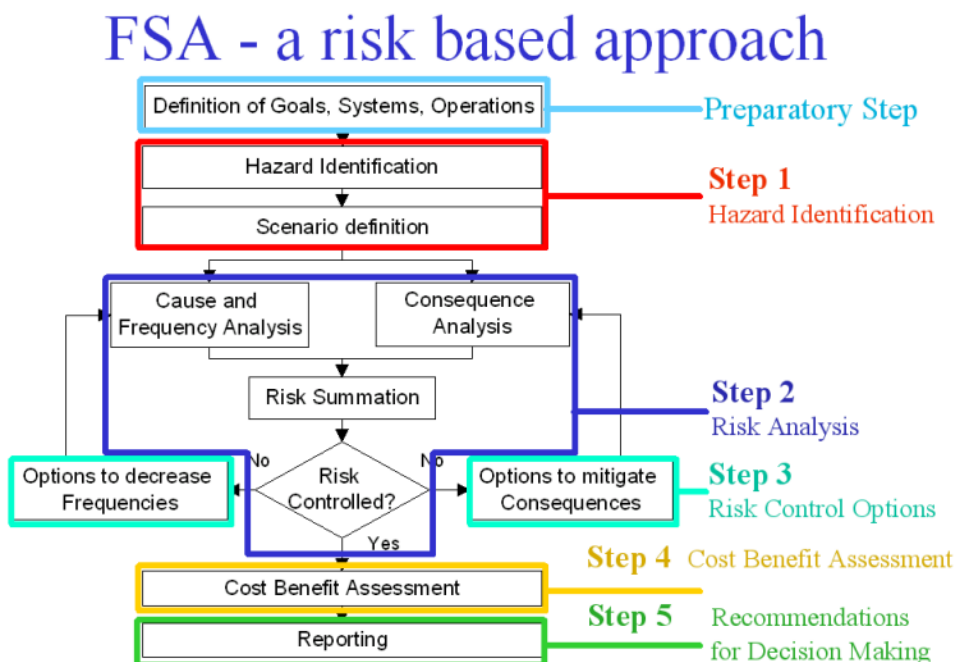


Fig. 1-2: FSA Flowchart -Source: IACS (2005)

In brief, the main objective of **Step 1 (Hazard Identification)** can be satisfied with a combination of creative and analytical exercises that aim to identify all relevant hazards. Note that most studies have extensively –if not exclusively- used historical data found in various casualty databases. The purpose of **Step 2 (Risk Analysis)** is the detailed investigation of the causes and consequences of the more important scenarios -that were identified in the previous step- in order to focus on high risk areas.

According to the FSA Guidelines, the purpose of **Step 3 (Risk Control Options)** is: *“to propose effective and practical Risk Control Options (RCOs) comprising the following four principal stages:*

- 1. focusing on risk areas needing control;*
- 2. identifying potential risk control measures (RCMs);*
- 3. evaluating the effectiveness of the RCMs in reducing risk by re-evaluating step 2;*
- and 4. grouping RCMs into practical regulatory options.”*

There is no doubt that this step is a very important one. The options that may be proposed for implementation are the results of this step. There is always the risk of leaving important reduction measures out of the following analysis but the most important point is that the effectiveness of risk control measures has to be evaluated by risk analysis. However, this is not the case in most FSA studies. Risk reduction that will be discussed in the next two Steps is mainly estimated by expert judgment and is usually a percentage of expected outcomes of the relative hazard.

The fourth Step of a Formal Safety Assessment (**Cost-benefit Assessment**) is to perform an assessment so as to pick which RCOs are most cost effective. According to the FSA guidelines, one stage of this Step is to *“estimate and compare the cost effectiveness of each option, in terms of the cost per unit risk reduction by dividing the net cost by the risk reduction achieved as a result of implementing the option”*. What we should clarify is that this step is not actually a cost benefit assessment in its traditional meaning but rather a cost effectiveness assessment which is a tool mainly used in health economics. In quite all FSA studies submitted to the IMO, the major outcome of this Step is the estimation of cost-effectiveness indices such as the Gross Cost of Averting a Fatality (GCAF) that is discussed in Section 3.1.

The final Step of FSA aims at giving recommendations to the decision makers for safety improvement taking into consideration the findings during all four previous steps. The RCOs that are being recommended should reduce current risk to the “desired level” and “be cost effective”.

1.5 The ‘desired’ and ‘acceptable’ risk

An important aim of risk reduction measures is “to assess the level of risk at which engaging in the business of shipping remains sustainable” for an individual company, the shipping industry and the society as a whole and the concept of risk tolerability is introduced for this purpose as a measure which describes the degree of risk acceptability (Li and Kullinane, 2003).

In general, tolerability can be divided into three levels based on the acceptability of risks. Figure 1 presents a conceptual schema showing these three levels, which is adapted from Li and Kullinane (2003). This is a more generic conceptualization than the one presented in Li and Kullinane (2003) in order to stress out that in practice the boundaries of these risk areas are not well defined. In addition, the consequence is presented in monetary units, which may well be the case for environmental risk. This concept is also valid for non-monetized consequences.

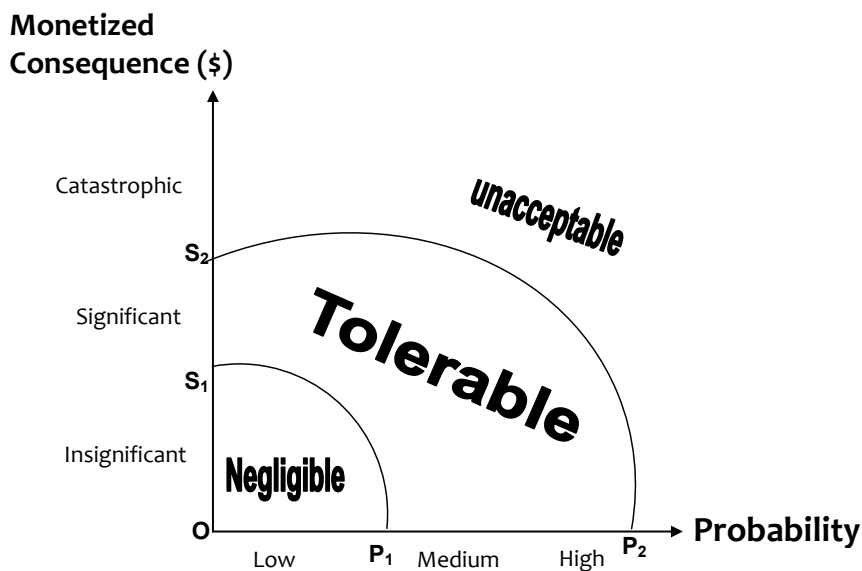


Fig. 1-3: Conceptualization of a hypothetical relationship between Tolerability, Probability and Severity - Adapted from Li and Kullinane (2003)

According to the Health and Safety Executive’s (HSE, United Kingdom) Framework for the tolerance of risk, there are three regions in which risk can fall into (HSE, 2001). One area is the negligible level (or acceptable risk level), where the particular combinations of the probability and severity (i.e. the risk) of a hazard fall within the area bounded by the points $OS_1O_1P_1O$ and no action is needed. Unacceptable Risk (for example resulting from high accident frequency and high number of fatalities) should either be forbidden or reduced at any cost. This is the area bounded by $OS_2O_2P_2O$.

Between these two regions, the tolerable risk region is defined. Within FSA, the notion of ‘desired level’ is linked with the so-called risk acceptance criteria and the ALARP principle. HSE (2001) defines this area as the ALARP (As Low As Reasonable Practicable) region. These regions according to the ALARP principle are illustrated in Figure 1-4 below. Note that, in general, there is distinction between ‘tolerable’ and ‘acceptable’ risk.

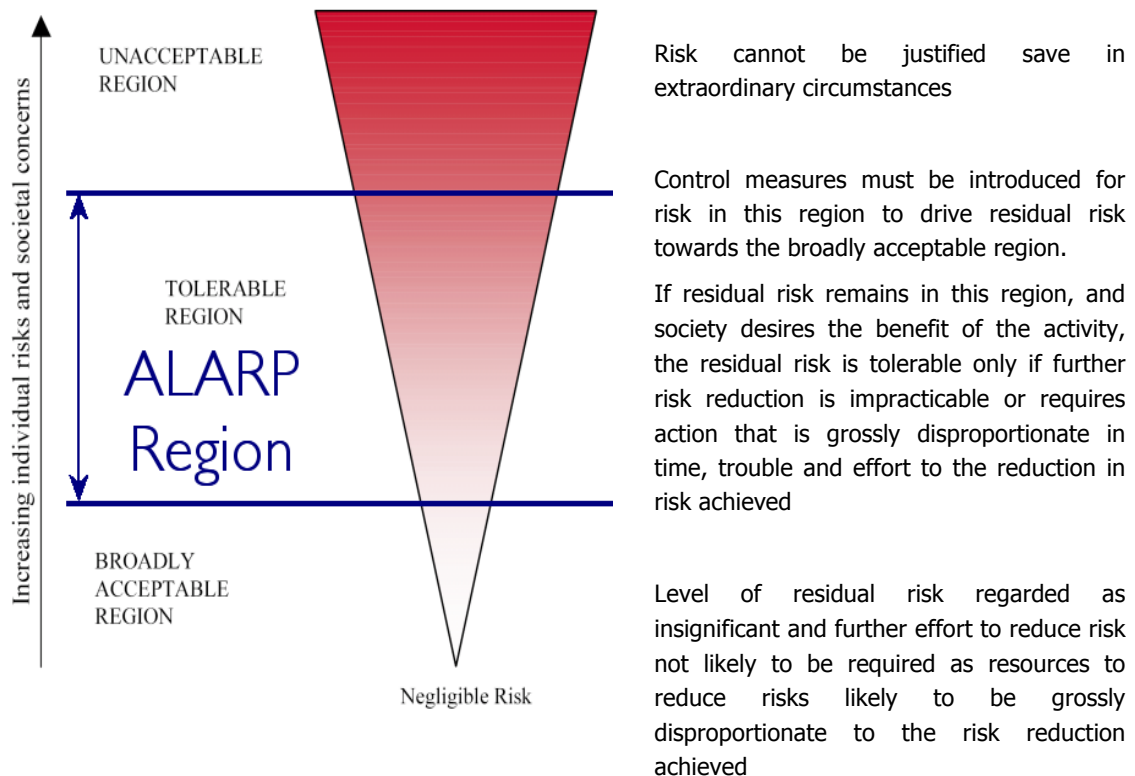


Fig. 1-4: The ALARP Concept – Source: Kontovas (2005) adapted from HSE (1999)

FSA guidelines provide no explicit Risk Acceptance Criteria. Currently decisions for human safety are based on those published by the UK Health & Safety Executive (HSE,1999) and we note that in the recently adopted amendments to the FSA guidelines (see Annex to doc. MSC 83/INF.2), it was made clear that all of these criteria are only indicative. Risks below the tolerable level but above the negligible risk (for crew members, passengers and third parties) should be made ALARP by adopting cost-effective Risk Control options (RCOs).

According to the FSA Guidelines, ‘Individual Risk’ is taken to be the risk of death and is determined for the maximally exposed individual. Starr (1969) observes that people’s willingness to accept risk is substantially greater for voluntary activities than for involuntary such. Indeed, in the indicative individual risk criteria presented in the FSA Guidelines, the maximum tolerable risk for crew members is 1/1,000 annually and for passengers and people ashore is 10 times less.

In extension, "societal concerns due to the occurrence of multiple fatalities in a single event is known as 'societal risk.'" (HSE, 2001). The purpose of societal risk acceptance criteria is to limit the risks from ships to society as a whole, and to local communities (such as ports) which may be affected by ship activities. In particular, societal risk acceptance criteria are used to limit the risks of catastrophes affecting many people at the same time, since society is concerned about such events (high consequence index). Usually, Societal Risk acceptance is based on the so-called FN-diagram. F-N diagram shows the relationship between the annual frequency F of accidents with N or more fatalities (Ball, 1998; HSE,2001). An F-N diagram is used to quantify societal risk as it accounts for large accidents as well as for small ones which enable us to express risk aversion. Risk aversion in F-N curves is used to express that, in general, society is less willing to accept one large accident with many fatalities than many accidents each with a small number of fatalities. The ALARP principle can also be used in this case.

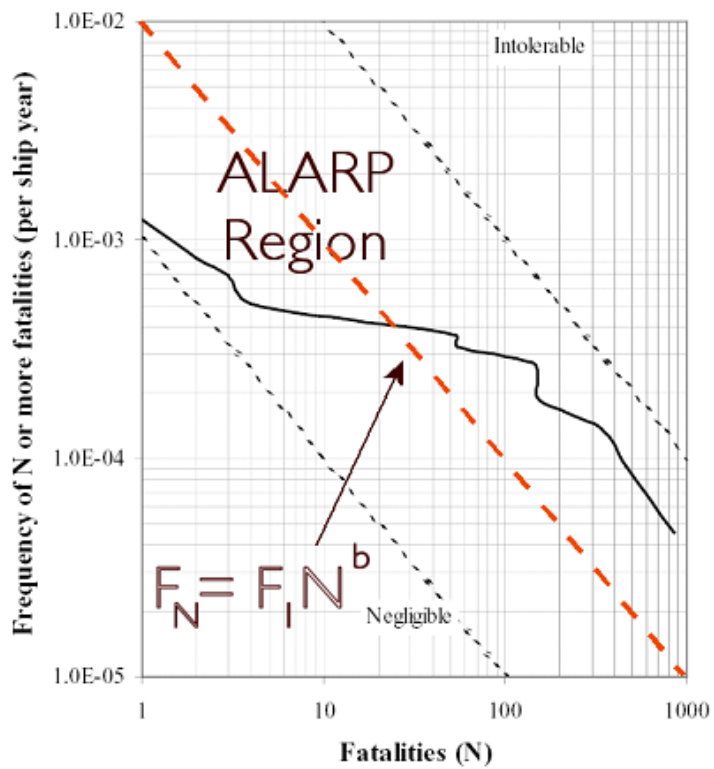


Fig 1-5: Typical F-N Diagram– Source: Kontovas (2005)

According to the ALARP principle, a risk reduction measure should be implemented provided it cannot be demonstrated that the costs are 'grossly disproportionate' relative to the gains obtained from reducing the risk further. To verify being ALARP, procedures mainly based on traditional cost–benefit analyses (that will be discussed in Chapter 2) can be used.

In short, the 'degree of disproportion' d of e.g. x , means that for a measure to be rejected, the costs should be more than x times larger than the benefits; and is defined as follows:

$$\text{disproportion ratio } d \leq \frac{\text{costs of a proposed risk control measure}}{\text{risk reduction benefits}}$$

There is neither an authoritative guidance on what ratio to employ, nor a formal algorithm for which factors to take into account and judgments should be made on a case by case basis. As 'rule of thumb' HSE (2009) uses as a starting point the submission to the 1987 Sizewell B Inquiry that a factor of up to 3 (i.e., costs three times larger than benefits) would apply for risks to workers; for low risks to members of the public a factor of 2 and for high risks a factor of 10.

The reader is referred to Johansen (2010) who has performed a throughout literature review of risk acceptance criteria. In addition, the author has performed an analysis of acceptance criteria used in FSA in Chapter 8 of his undergraduate diploma thesis, see Kontovas (2005).

It should be stressed out that most risk acceptance criteria concerning risk to human life avoid monetizing this risk and use non-monetary units such as the amount of fatalities averted. In all other cases, the risk can be easily monetized and risk tolerability can be approached by the accepted monetized risk. In this case, the generic conceptualization where the consequence is monetized as presented in Fig. 1-4 can be used.

Furthermore, in such cases it may be more interesting to investigate the optimal level of safety and environmental pollution, see next Chapter and one possibility may be to define this optimal level as the boundary between the 'acceptable' and 'unacceptable' region. To that extent, the acceptable level of risk can also be formulated as an economic decision problem where the expenditure for a safer and more environmental friendly state is equated with the gain made by the decreasing present value of the risk. The acceptable risk can then be transformed to a limit for the expected economic damage in monetary units per time unit (e.g. \$ per year, or \$ per ship year). This issue is out of the scope of this dissertation but definitely deserves some investigation in the future.

1.6 Economic Appraisal within Public Policy & Risk Assessment

It is evident that a risk-free shipping industry is not possible. Section 2.2 will indeed present that there is an optimal societal (and private) non-zero level of activity both for human safety and environmental pollution. In addition, currently, most studies support the fact that individual and societal risk for most ship types are in the ALARP region, see Skjong (2002a) and all FSA submitted to the IMO by the SAFEDOR project (SAFEDOR, 2008). However, more analysis deems necessary as one single accident may move the entire fleet of a specific ship type to the unacceptable region. It seems that the most crucial process is that of weighting the cost of risk control measure with its benefits.

There are a couple of techniques that weigh up the costs of an action against the benefits that it provides. In general, these techniques are applied before the action is taken (*ex ante*) to decide what is to be done or choose among a set of possible actions -this is called Economic appraisal- and after the action to evaluate the effects of its implementation -this is referred to as economic evaluation. These techniques can be part of other frameworks (for example a risk assessment) or standalone procedures.

These methods are also widely used in public policy, see next Chapter. For example, the United States and the United Kingdom have detailed guidance on how to conduct relevant analyses. However, probably the main disadvantage of CBA is that it seeks monetization of all the effects. Some people feel that it is unethical to place a monetary value on health or mortality risk changes because it seems that CBA places a value on human life (Krupnick, 2004). Furthermore, although there is nothing wrong with its theoretical background, apart these ethical issues, one main problem (if this can be considered one) is that CBA needs a lot of resources and time. One way to avoid potential ethical issues and to arrive at faster results is to use the so-called cost effectiveness analysis (CEA) which compares the costs of alternative ways of producing the same or similar outputs. A CEA looks at the amount of cost per unit of effect – that is C/E , where C are costs of a procedure and E the relevant consequences (effects) (Brent, 2003)

These main techniques for economic appraisal will be presented in the following Chapter. What has been said until now is sufficient to place these techniques within risk assessment although as it was discussed before these techniques can also be used in a standalone form. The objective of CBA in a risk assessment context is to identify cost-effective risk reduction measures. Note that, as it will be analyzed in the next

Section, currently within Formal Safety Assessment the main tool for economic appraisal is Cost Effectiveness Analysis and not Cost Benefit Analysis. In any case, the way of placing these techniques into a risk assessment framework is almost the same and is conceptualized in Figure 1-6.

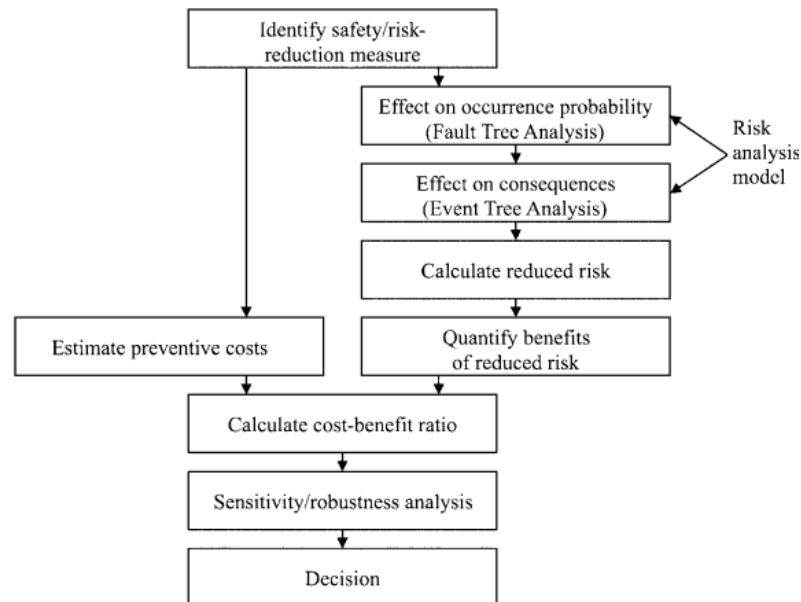


Fig. 1-6: CBA in a risk assessment context – Source: Kristiansen (2005)

As a conclusion of this introductory chapter, this dissertation deals with the rather difficult issue of monetizing risk and evaluating alternatives to reduce this risk. The advocacy of an approach based on cost–benefit analysis in the regulation of environmental, health and safety contexts is nothing new (Arrow, 1997). However, little has been written on how to apply valuation techniques in applied maritime risk assessment, especially for not health related risks. To that extent, this thesis presents the work performed by the author and other members of the Laboratory for Maritime Transport within the International Maritime Organization as regards risks related to oil spills. In addition, there has been no FSA study that assesses reduction measures to reduce ship air emissions. Finally, to the author’s knowledge there has never been any practical example of using non-linear damage functions within FSA (except some cases for oil spill pollution risks); a step which will definitely lead in engaging more of Cost Benefit Analysis. These are indeed some of the strong points of this works.

By all the above, this dissertation couples the practical way of thinking with the theoretical background provided by economic theories, identifies possible ways to monetize and evaluate risk in such a practical way that can be easily incorporated in risk assessment frameworks and as a result contributes to the better understanding of the relevant economic risk for an efficient risk-informed decision making.

2

VALUATION AND EVALUATION TECHNIQUES

Valuation techniques, as the word implies, are used to attach a “value” to the benefits of a certain risk reduction measure. This is equivalent to monetizing, in our case, the potential risk. The use of evaluation techniques is to weigh the benefits of an action against the costs. This is the main criterion that we all humans apply everyday when making decisions. We evaluate alternatives or decide whether we should go ahead with a particular decision or just stay with the current situation. In our everyday decision-making we impose criteria based on our perceived utility of an action and then compare it with that of the alternatives. Similarly, when an action that imposes risk in human life is identified then measures to reduce these risks should be sought constrained of course by the budget.

Policy makers apply the same criterion where costs and benefits are judged from a wider perspective, that of society as a whole. It is clear that there are a lot of difficult points in doing so; see Hanley and Barbier (2009). For example, what do we mean by ‘society’ and what should we include as the ‘benefits’ or the ‘costs’ ? How do we ‘value’ these and how we do this for actions (measures, policies or projects) whose costs and benefits stretch far into the future, for example those related to Climate Change? Some answers can be are provided by evaluation techniques such as Cost-Benefit Analysis (CBA) and Cost-Effectiveness Analysis (CEA). These tools can be used for individual decisions but also for decisions in the private sector (firms) and regulators (governments).

Economic Appraisal within Central Government

As discussed in the introductory Chapter, the use of these techniques is mandatory in some countries. In the United States, CBA within policy making has a long history. In generally, the implementation of CBA is associated with the Reagan Administration and Executive Order 12291 which required agencies to produce regulatory impact analyses (RIA) on regulations with a likely annual impact of \$100 million or more. US President Clinton issued Executive Order 12866 in 1993 to govern regulatory review requiring that benefits of a regulation justify the regulation’s cost rather than the formulation in the Reagan Executive Order which required that benefits exceed the

costs (Shapiro, 2010). Under Executive Order 12866, U.S. Government agencies are required, to the extent permitted by law, *"to assess both the costs and the benefits of the intended regulation and, recognizing that some costs and benefits are difficult to quantify, propose or adopt a regulation only upon a reasoned determination that the benefits of the intended regulation justify its costs."* On January 18, 2011, President Obama signed Executive Order 13563, 'Improving Regulation and Regulatory Review', re-affirming US President Clinton's directive.

CBA and CEA are used in economic appraisal in many countries, without although being mandatory. For example, before the European Commission (EC) proposes new initiatives it assesses the potential economic, social and environmental consequences by performing Impact assessments (IA) that may include a cost-benefit or cost-effectiveness analysis. CBA and CEA are also widely used in the United Kingdom. The so-called "Green Book", the HM Treasury's guidance for Central Government, sets out a framework for the appraisal and evaluation of all policies, programs and projects.

However, in the UK, the most common form of analysis in government especially to assess health effects is cost effectiveness analysis (CEA), where the costs of alternative ways of providing similar kinds of output are compared and the effect in health is not monetized, as in CBA, but measures such as the years of life gained or premature deaths averted and others are used. Actually, it is probably true that worldwide cost-effectiveness analysis has emerged as a favored analytical technique for economic evaluation in health care.

For a number of years, the United Kingdom, Canada, Australia, and other countries have explicitly incorporated cost-effectiveness considerations, for example, in making coverage and pricing decisions drugs and medical treatments. The U.K. Department of Health has commissioned the National Institute for Health and Clinical Excellence (NICE) to make recommendations on the basis of both clinical effectiveness and cost-effectiveness. Decisions made by the National Institute for Health and Clinical Excellence (NICE) about whether the UK National Health Service (NHS) should fund treatments are also based on cost effectiveness.

In the United States, since the 1960s and early 1970s, health officials at the U.S. Department of Health, Education, and Welfare began applying CEA to a variety of health problems, including kidney disease and maternal and child health programs (Klarman, 1968). However it is argued that there is some resistance in widely accepting CEA in the United States (Neumann, 2005).

Finally, in a more international framework, The CHOICE (CHOosing Interventions that are Cost-Effective) project is a major WHO (World Health Organization) initiative developed in 1998 "with the objective of providing policy makers with the evidence for deciding on the interventions and programs which maximize health for the available resources" (WHO, 2003).

2.1 The Microeconomic foundations of valuation and evaluation

This section briefly covers the basic theoretical framework especially of cost–benefit analysis with an objective is to give a clearer picture of the theory behind the concepts of individual and social welfare and the decision criteria based on the net present value. Most of the material (including notation) in this Section is based on de Rus (2010). More on the conceptual basis of cost benefit analysis in welfare economics can be found in Boardman et al. (2001;2010), Just et al.(2004), Mishan and Quah (2007) and de Rus (2010).

To start with, *positive economics* is the branch of economics that is concerned with understanding and predicting economic behavior. On the other hand, welfare economics is concerned with what 'ought' to be. Welfare economics belongs to *normative economics* as it focuses on using resources optimally to achieve the maximum well-being for the individuals in society and is based is a set of ethical assumptions. Two of these ethical assumptions are widely accepted as providing the foundations of the applied welfare economics and policy evaluation. The first one is that "the welfare status of society must be judged solely by the members of society, which recognizes the traditional emphasis on the importance of the individual in Western society" also called the *fundamental ethical postulate* or the principle of *individualism* (Quirk and Saposnik 1968) according to which the only thing that counts is the preferences of the members of society. The second proposition is that society is better off if any member of society is made better off without making anyone else worse off also known as the Pareto Principle, which will be analyzed later on.

The Utility Function

Let there be N people in our society indexed by $i=1,..,N$ and assume that there is one composite good, called \mathbf{x} . Let $x=(x_1,..,x_N)$ represent the individual consumption of the material good and e represent the quality of the environment which is assumed to be the same for all. Note that this is valid for the case of CO2 emissions which is a global problem but not true for regional pollutants such as SOx and oil spills that usually affect a specific area; however, this is true for the subset of the 'local society'. Kolstad (2011) defines the utility or well-being obtained by individual i as $U(\mathbf{x},e)$. One may

assume that the individual cares only about the consumption of x_i but there is no loss of generality by assuming the previous. This is based on altruism which means that the individual considers the others (and possibly future generations) when making decisions.

All individuals have preference on the quantity of consumption which is expressed by their utility function. That is, if the individual prefers a to b , then $U(a,e)$ is greater than $U(b,e)$. The reverse is also true. This above can also be extended for a bundle of goods.

The greatest difficulty in welfare economics is that economic 'welfare' is not an observable variable and, therefore, the economic welfare status of an individual is formally represented by his or her utility level. However, one cannot measure the changes of welfare in utility. The associated measurement problem was addressed by Hicks (1943) who suggested that alternative money measures of welfare, while not directly related to utility gains and losses, can be given willingness-to-pay (WTP) interpretations (Hicks, 1956). For example, one could measure how much money each individual is willing to pay for a particular policy change, which may be negative for some individuals, and then observe whether the total is positive or negative. The notion of WTP and the associated compensation criteria are key concepts and form the foundation of applied welfare economics and, in extension, of Cost Benefit Analysis.

To that extent, welfare economics is not about money; it is about individual and social welfare. Money is central in financial analysis but only instrumental in economic appraisal since money is just a common unit to express the costs and benefits. Probably the monetization of the effects especially of human health is one reason for the criticism of welfare economics.

The Social Welfare Function

By analogy, the preferences of the society can be represented by a similar utility function. The assumption that the social welfare function W is determined by the utilities of all individuals has been the *fundamental ethical postulate* by Samuelson (1947) and is the so-called *Bergsonian welfare function* after Abram Bergson (1938), who first used it. Early students of the utilitarian school (Bentham, 1961) believed that changes in happiness should simply be added over individuals. The Benthamite function (after Bentham) is the following:

$$W=U_1+U_2+\dots+U_N$$

There is a number of examples of social welfare function. Another very well known function is based on Rawls's (1971) *Theory of Justice* which presents the principle that the policy should be evaluated by the welfare of the most miserable person in society, which implies the a social function (also known as the Rawlsian) in the following form :

$$W=\min(U_1,U_2,\dots,U_N)$$

It should be noticed that there is no generally acceptable or objective way to make interpersonal comparisons of utility and, therefore, no commonly accepted social welfare function exists.

Extra-welfarism

In an extension of welfarism, extra-welfarists (e.g. Culyer, 1991) argue that utility is not the only relevant argument, or indeed even the most important argument, in the social welfare function. For example, Sen (1980) is arguing that a focus on individual utility is too narrow and ought to be replaced by "*a broader perspective to take into account the quality of utility and of people's capabilities rather than exclusively of the emotional reaction (i.e. utility) of individuals to the possession of goods or capabilities*" (Brouwer et al., 2008).

CEA, like CBA, values all impacts ('costs') in dollar terms, or at least tries to; but it values the major effect (f.e health impacts) on some non-monetary 'effectiveness' scale. The question of monetizing or not the effect is often seen in the health policy literature as a choice between 'welfarism' and 'extrawelfarism'. Welfarism favours CBA, extra-welfarism favours CEA – or so it is often assumed. This assumption is probably mistaken but it is true that CEA should be seen by extra-welfarists as little better than CBA. To adequately develop the point, we would need to get clear about the content of extra-welfarism, which is, however, out of the scope of this work. Brouwer et al. (2008) is a good reading for those interesting in the topic.

In short, it is probably true that CBA has a strong underlying basis is neo-classical economic theory; stronger than the basis of CEA which is often seen to be rather weak. For the economic foundations of CEA the reader is referred to Brouwer and Koopmanschap (2008).

Social Choice Mechanisms

Now, given such a social function one may feel that there is an ideal way of making social choices. It may surprise those that are not familiar with the theory of Social Choice than Kenneth Arrow (1951) presented the so-called *Arrow's Impossibility Theory* which, unfortunately, demonstrates that it is extremely difficult to derive a complete and consistent choice rule exclusively from individual preferences. This

implies, for example, that no voting system can convert the ranked preferences of individuals into a complete and transitive ranking while also meeting a certain set of criteria (including the Pareto efficiency and non-dictatorship). So, if we cannot use individual preferences to derive a social ranking, what are then the second-best choice rules?

Vilfredo Pareto (1896) introduced a welfare criterion, the Pareto optimum, which became a foundational concept in welfare theory. Allocations are said to be Pareto optimal if *“no other feasible allocation could benefit at least one person without any deleterious effects on some other person.”*

Allocations that do not satisfy this definition are suboptimal such as that some people can gain net benefits without the rearrangement causing anyone else to lose net benefits. In that case gainers could use a portion of their gains to compensate the losers sufficiently to ensure they were at least as well off as they were prior to the reallocation. It is obvious that very few policies have no losers.

Therefore, Kaldor (1939,) suggested that where a policy led to an increase in aggregate real income it was desirable because in such cases *“it is possible to make everybody better off than before, or at any rate to make some people better off without making anybody worse off.”*

Thus, the explicit decision rule within CBA is based on what is known as the Kaldor-Hicks criterion (also called the potential Pareto improvement) : *“A policy should be adopted if and only if those who will gain could fully compensate those who will lose and still be better off”* (Boardman et al., 2001). In theory there needs to be no compensation, the compensation is hypothetical. However, if losers are compensated this will result in a Pareto improvement. Therefore, Kaldor-Hicks criterion provides the basis for the CBA rule which demands that the regulator adopts those policies that have positive net benefits (Boardman et al., 2010; de Rus, 2010).

Equity and distributional effects

Every economic process should be considered in two separate lights (a) the optimal quantity of production and (b) who gets the benefits, or the "distribution". Almost without exception, the great works in economics have focused on these aspects of the operation of the economy which are often termed as 'efficiency' and 'equity', respectively. Economic efficiency (which will be analyzed in the next Section) has to do with producing and facilitating as much consumption as possible with available resources, whereas *equity* has to do with how equitably goods are distributed among individuals.

Equity has to do with the fact that it is often assumed that individuals have comparable utility and that the marginal utility of money is (almost) constant. In this meaning, equity is somewhat different than equality and is taken to mean that there is some fairness on the distribution of benefits and costs. To that extent, equity is a probably more a problem for the moral philosopher (and definitely the decision maker) rather than the economics or the engineer who performs a CBA. Note, that it is not straightforward to do the same within CEA.

In the past, equity has been ignored in decision-making process, sometimes in an attempt to give an objective advice. However, this is a very important issue, which has to be addressed, but there is no straightforward way to do it within Cost Benefit Analysis. Take for example the case of an oil pollution where the ship-owner is responsible for the damage to the society and we have to address a related risk reduction regulation. Should we include the property cost of the lost oil in the damages (and, thus, the benefit of averting) of such an oil spill? Some people may say that the polluter is the 'criminal' and it should be only the damage to the society that counts. Or take for example a project that affects both children (and young people) and adults. Probably the calculus of willingness to pay (or willingness to accept) for risk reductions cannot be the same because children may have no income to allocate between goods. Similarly, CBA risks under-weighting the interests of the poor (or, sometimes, over-weighting them) simply because the utility of money is greater for the poor.

Indeed many agencies are now required to consider the distributional impacts of costs and benefits as part of an economic analysis; e.g, the U.S. EPA (2010) guidelines on distributional issues in Chapter 10 of the "Guidelines for Preparing Economic Analysis." According to the EPA, distributional analysis assesses the impact of regulations across different subpopulations. Although there exists no formal guidance until now, the EPA Guidelines mention the need to contact distributional analyses. Distributional analysis is either an economic impact analysis or an equity analysis. Economic impact analysis focuses on the determination of who gains and who loses from a given policy and equity analysis examines impacts on disadvantaged groups (e.g. minority and low-income groups) or subpopulations (e.g. tribal populations).

On the other hand, in the United Kingdom, Appendix 5 of the "Green Book" (HM Treasury guidance for Central Government) proposes the use of distributional weights based on the individual's relative prosperity (f.e income) to adjust explicitly for distributional impacts. To that extent, benefits accruing to individuals in a lower quantile would be weighted more heavily than those that accrue to individuals in

higher quantiles and, similarly, the costs would be weighted more heavily for individuals in lower quantiles. However, it is noticed that this may be contrary to discrimination laws and this is also an issue to be taken into account.

To conclude, we stress out that CBA should start by identifying the groups on whom the costs and benefits of the action will fall and the fairness among gainers and losers should be thoroughly addressed. Clearly, this issue should be addressed more in risk assessment and CBA. The reader is referred to Loomis (2011) who outlines several approaches for incorporating distribution and equity into benefit–cost analysis.

2.2 Efficiency: The optimal level of safety, health and environmental protection

Criteria for evaluating policies are based on their ability to achieve efficient and cost-effective risk reductions. 'Efficiency' is the balance between abatement costs and damages (Field, 2003). **An efficient policy is one that moves the society to, or near to, the point where marginal abatement costs and marginal damages are equal. That is the level of the optimal damage.** Since that environmental damages cannot be measured accurately, the cost-effectiveness criterion is the most useful to be employed. As described in Field (2003), a policy is cost-effective if *“it produces the maximum environmental improvement possible for the resources being expended or, equivalently, it achieves a given amount of environmental improvement at the least possible cost”*.

It is important to elaborate a little more. Generally, the marginal damage caused by a unit of pollution increases with the amount emitted. This means that when small amounts of the pollutant are emitted, the incremental damage is quite small and when these amounts become larger the marginal unit can cause more damage. This is easy to understand, for example, in the case of oil pollution since the ocean can tolerate (i.e. dissolve) small quantities of oil. On the other hand, marginal control costs commonly increase with the amount controlled since it becomes more and more costly to remove each extra unit added.

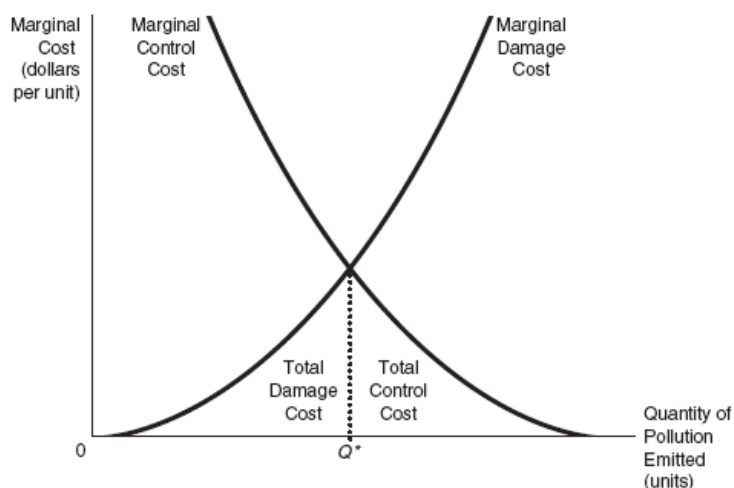


Fig. 2-1: Illustration of efficient allocation - Source: Tietenberg (2011)

Figure 2-1 (based on Tietenberg, 2011) illustrates the typical shape of the relevant curves. In the absence of externalities and other market failures (briefly discussed in Section 2.2.3), the efficient allocation which is represented by Q^* occurs precisely at the point at which the damage caused by the marginal unit of pollution is exactly equal to the marginal cost of avoiding it. At this point, total damages are not equal to the total benefit (i.e. the cost of avoiding the damage). It is obvious that the previous formulation is equivalent to the net benefit formulation. Since the benefit equals the reduction in damage, another way of stating the previous proposition is that marginal benefit must equal marginal cost. That is, of course, the case when net benefits are maximized.

In addition, since the net benefits are maximized by an efficient allocation, it is not possible to increase the net benefit by rearranging the allocation. Without an increase in the net benefit, it is impossible for the gainers to compensate the losers sufficiently; the gains to the gainers would necessarily be smaller than the losses to the losers. This means that efficient allocations are also Pareto optimal.

As Fig. 2-1 suggests under the conditions presented, the optimal level of pollution is not zero. This is also true for the optimal level of any kind of risk, including risk to human health and life. The existence of a risk does not in general imply the need for regulatory action. For instance, people can be compensated to stand such a risk. In the case of job safety, perceived risks of hazards lead to compensating differentials for this risk and workers receive wage compensation sufficient to make them willing to bear the relevant risk (Viscusi, 1993). Obviously in this ideal market, the greater the risk the greater the compensation received. That means, the wage of a master in an LNG carrier should be greater than that of a master in a general cargo and much less. This is

related to the previous discussion on the acceptable risk. The level of risk that is acceptable for a crew member is higher than the acceptable risk for a passenger, just because the passenger is not compensated for accepting the extra risk. So, in a fully functioning market, workers receive wage compensation that is sufficient to make them willing to bear the risk; which is how "*the health risk is internalized into the market decision*" (Viscusi,2002).

The idea of an optimal, non-zero, level of pollution or human damage may sound indeed strange, and unethical, to most people. Obviously, in environmental economics, the notion of optimal level of pollution is central and indeed not that strange although there are still people that may think that anything but zero pollution is unethical. After all, if you still find that the optimal level of damage to human life being not zero disturbing, remember that you probably confront this principle every single day. Take the damage caused by automobile accidents, for example. A lot of people die (worldwide every day), yet we do not reduce that damage to zero because the cost of doing so would be too high. And we do know how to avoid all such accidents: we eliminate cars! But this is obviously not what society wants.

In the following sections, we briefly analyze how the above rationale can be applied in health and safety but also to the environment and we present some interesting issues such as the value of preventing fatalities and pollution, the limited compensation liability and the notion of "externality".

2.2.1 Optimal level of 'risk' to humans

Viscusi (1993) discusses the fundamental economic approach to worker safety as sketched by Adam Smith within the Compensating Wage Differential Theory, by stating that it is primarily the risk-dollar trade-offs of the workers that will determine the safety decision by the firm. For the compensating differential model to be applied workers should be aware of the relevant risk they face and be able to make sound and rational decisions based on this knowledge. To that extent, Figure 2-2 illustrates the marginal value of the safety curve which is a down-sloping curve since the initial increments for safety have the greatest value as other types of "economic goods". On the other hand, the firm can provide increased safety but in an increasing incremental cost and obviously this cost goes to infinite to provide a nearly risk-free work environment. It is, then, the price of safety set by the workers that determine where along this marginal cost curve the firm will stop installing equipment to reduce safety risk.

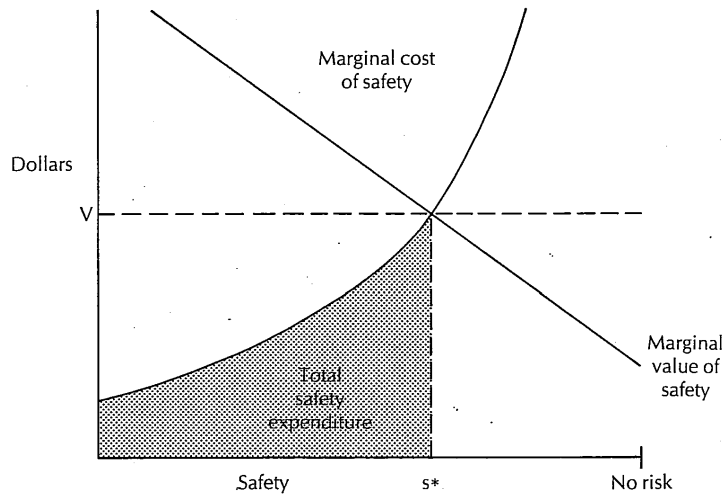


Fig. 2-2: Determination of Optimal Level of Safety – Source: Viscusi (1993)

At this optimal level of safety s^* , workers would have been willing to pay $\$V$ per expected accident in order to avoid such risk. In a perfect market this willingness to pay to avoid such expected accidents should be equal to the amount of money that the workers are willing to accept to bear the relevant risk. We may call this the per unit “price” of human safety. This is one of the values that can be used as the “Value of Statistical Life” also called the “Value to Prevent a Fatality” or more recently the “Value of Mortality Risk”. This issue will be discussed in Chapter 3.

Note, that here is substantial literature evaluating tradeoffs between money and fatality risks. These values in turn serve as estimates of the value of a statistical life, what is also known as the labor market estimates of the 'Value of Life'. Viscusi and Aldy (2002) review more than 60 studies of mortality risk premiums from ten countries and approximately 40 studies that present estimates of injury risk premiums. After all, it was Adam Smith (1776) who noted in ‘The Wealth of Nations’ that: *“The wages of labour vary with the ease or hardship, the cleanliness or dirtiness, the honourableness or dishonourableness of the employment”* (p. 112).

It should be stressed out that this conceptualization which is standard in economic theory may be applied not only for individuals and firms but also for the society as a whole. This conceptualization, which is standard in public welfare economics, can provide the social optimum level of safety at the point at which the marginal utility associated with the benefits of regulations to reduce human risk is equal to the marginal (dis)utility associated with the costs. Figure 2-3 illustrates the above concept based on Li and Kullinane (2003) as it can be applied in maritime safety and shows the optimum level of safety achievable by regulation to reduce maritime risk.

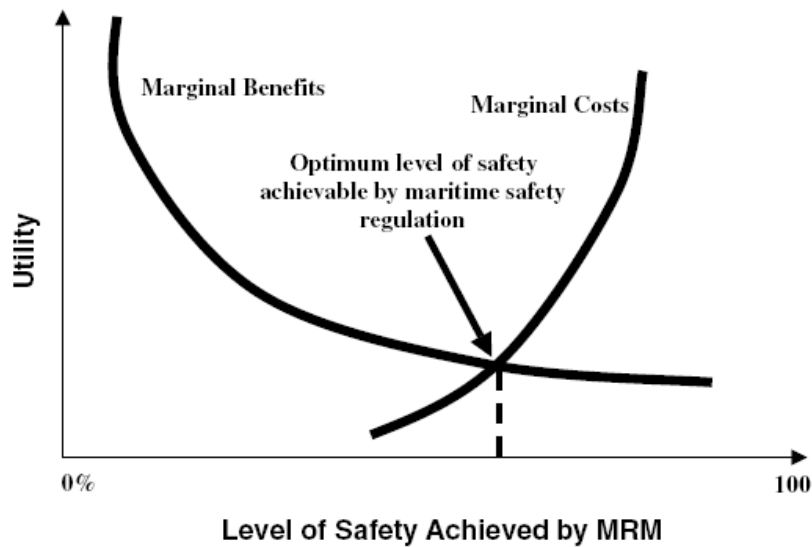


Fig. 2-3: Marginal costs and benefits of maritime safety regulation – Source: Li and Kullinane (2003)

However, in reality markets are not perfect. There are indeed situations where market forces might not operate effectively to internalize the risk. It was noted above that for this model to be fully applicable the society should be fully informed about the relevant risks. In addition, limitation of liability (that will be presented in the case of oil pollution) and market “externalities” (that will be presented in the case of air emissions) are, to name a few, the reason for not being able to achieve this optimal level.

Note, that limitation of liability also exists in the case of human safety. The shipping industry is a truly global industry, and such limitation regime have been adopted by most maritime nations in order to maintain a reliable sea trade and spread maritime risks among shipowners and other parties. For example, Li and Kullinane (2003) name the two international conventions, that is, the International Convention Relating to the Limitation of the Liability of Owners of Sea-going Ships 1957 (1957 Convention) and the Convention on the Limitation of Liability for Maritime Claims 1976 (1976 Convention) and its 1996 Protocol.

2.2.2 Optimal level of oil pollution

The following figure illustrates the incentive of owner to take precaution, or similarly the efficient point in implementing a regulation to prevent oil pollution. The figure, which is based on Tietenberg (1996;2011), presents the case of water pollution from oil spills. By forcing the vessel owner (or the Compensation Fund) to pay for the costs of an oil spill this creates the incentive for the owner to exercise care and for the regulator to implement control options to mitigate the pollution risk.

Furthermore, the figure below illustrates the major characteristic of the legal system through liability law. The efficient point as described above given unlimited liability is Q^* . As described in Section 3, admissible claims cannot be paid in full, especially in the case of large spills, since the total compensation paid is limited. That is true for most compensation systems including IOPC Funds but is also the case of the US Superfund. Thus, with limited liability the expected penalty is reduced and the level of precaution lowers, then the efficiency point is depicted as Q .

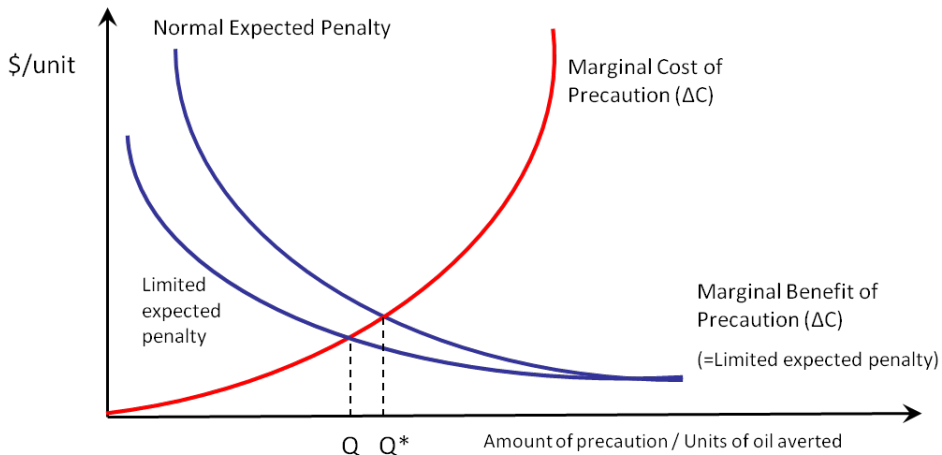


Fig. 2-4: Oil Spill Liability – Adapted from Tietenberg(1996).

2.2.3 Optimal Level of Pollution from Air Emissions

In the following figure, MC is a typical marginal cost curve, MD a typical marginal damage curve and x^* is the point of “efficient” pollution. Note that when talking about prevention of emissions, the benefits of preventing pollution are equal to the damage costs that would occur if the same amount of gases was released to the environment.

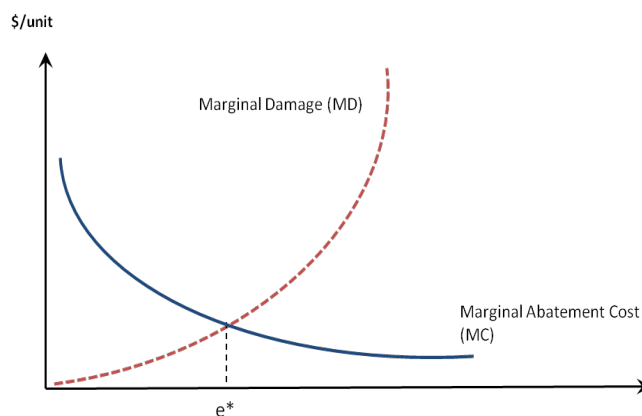


Fig.2-5: The efficient level of emissions

Fig. 2-5 represents the shape of the typical marginal cost and damage functions, and e^* is the socially optimal level of emissions. In equilibrium theory, it is worth reducing CO_2 emissions up to the point where the marginal benefits of reduction are equal to their marginal cost.

The case of air pollution is an excellent example to present the notion of market failures; that is cases that the market fails to operate to operate efficiently. Note, that market failures also happen in other kind of markets. It is to no surprise that market failures also happen when determining the optimal level of human safety. One case may be the lack of information about the risk that a specific activity poses to the work. The worker by being ignorant about the full amount of risk that he or she faces cannot make an optimal decision on the appropriate wage that compensates for the full risk.

Going back to air emissions, an *externality* is defined as the case “where an action of one economic agent affects the utility or production possibilities of another in a way that is not reflected in the marketplace” (Tietenberg, 2011). This is exactly what happens when the producer of a good emits some kind of emissions (due to the production). This is the case of negative externalities, shortly that is the case involving something “bad” for the society.

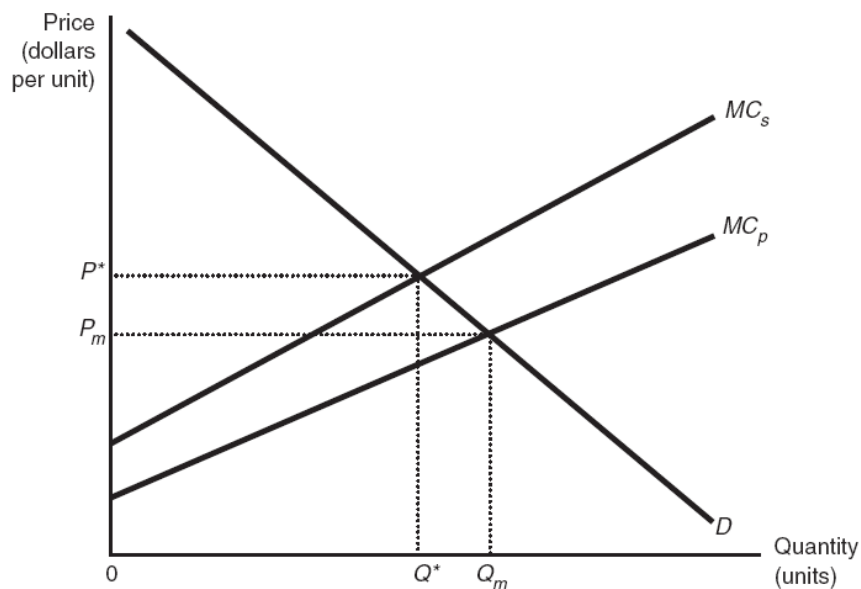


Fig. 2-6: Private and social efficiency in the presence of a negative externality Source: Tietenberg(2011).

The effect of external costs is illustrated in Fig. 2-6. The production of a "good" (could be maritime transportation) involves producing pollution as well. The demand for the good is shown by the demand curve D and the private marginal cost of producing the good (in the absence of pollution control) is MC_p . MC_s depicts the social marginal cost which includes the private cost and the damage of the pollution. In the absence of any control on emissions (which is something that the regulators should take care of) the firm produces Q_m , which is the optimal level of production that maximizes the producer's surplus. However, when the cost to the society is taken into account then the optimum level of production is at Q^* should be less than the private optimum Q_m .

There are a couple of ways to internalize this externality some of which are getting high attention lately in order to address the problem of Climate Change. From the environmental economics' point of view, as the Stern review states, climate change is the greatest and widest-ranging market failure ever seen, presenting a unique challenge for economics. In environmental policy-making, policies are often classified in market-based, command-and-control and voluntary instruments. We shall briefly discuss the market based policies; the most well-known by far are related to emission charges and trading schemes. The reader is referred to basic Environmental Economic textbooks such as Field and Field (2010) and Tietenberg (2011) for more. The author has also presented emission reduction related policies in Kontovas and Psaraftis (2011).

Emission Taxes

More precisely, charges are emission taxes or fees levied on the discharges. Most economists favor emissions taxes following the idea of Pigou (1920) that by charging for every unit of emissions released firms tend to reduce their emissions. Note that obtaining all necessary information to impose the ideal tax is quite costly and, in practice, regulators determine the charge using and trial-and-error process. Under an charge regime, the government will not only, in the long term, make the firm reduce its emissions to some desired level, but also be able to generate tax revenue that could be used to further clean up the environment or for any other social objective. The biggest problem of such a system is the effective monitoring.

Tradable emission permits

Clearly, there are market failures involved in most environmental problems especially when talking about public "goods" such as clean water (that is polluted by oil spills) and clean atmosphere that is polluted by ship emissions typically due to the lack of property rights. By being "public" obviously no one has property rights or actually one may say that these belong to the society and not to any subgroup of individuals. Without analyzing it further, these kind of problems may be solved by assigning property rights that can create a market for the externality. Coase (1960) was first to show *Pareto optimality* is attained when such markets are created. This is the so-called Coase Theorem, the core of which is that if a polluter owns the right to pollute, the pollutee maybe willing to pay the polluter to either reduce or stop pollution. If the pollutee owns the right to no pollution, a potential polluter may want buy from the polutee the right to pollute (Just et al., 2005).

To that extent, tradable emission permits allow the voluntary transfer of the right to emit from one firm to another. In this system firms are allocated a number of emission permits and are entitled to emit one unit per permit but these permits are transferable. A market for these permits will eventually develop and firms that can reduce emissions at a low cost may prefer to sell its permit to a firm that can reduce pollution only at a high cost. 'Cap-and-trade' programs work a little bit different since the first step is to make a centralized decision on the aggregate quantity of total emissions. The permits are then distributed among the emitters and traded in the market and will flow from firms with lower marginal abatement cost to those with higher cost. Thus, there will be an incentive for them to look for ways to reduce emissions. Those emitters who can reduce their emissions more cheaply can and will sell the extra allowances to others who otherwise have to pay more to comply and because of this, a cap-and-trade system helps assure that we can achieve an overall cap at a low cost.

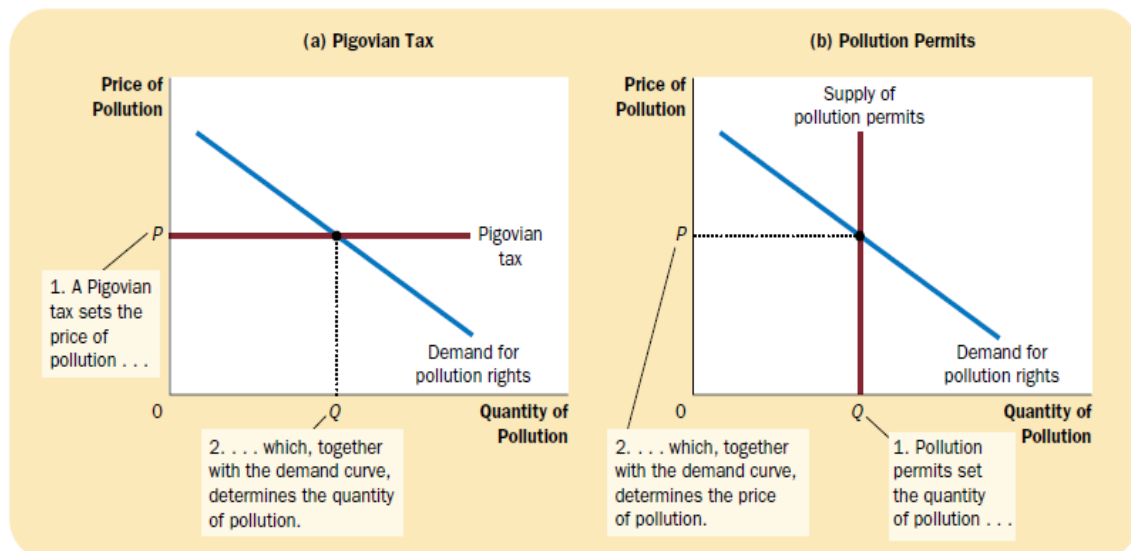


Fig. 2-7 :The Equivalence of Pigovian Taxes and Pollution Permits - Source: Mankiw (2006)

One may notice that these two systems lead to equivalent results in the long term but with different uncertainty for the outcome. The basic difference is illustrated in Fig. 2-7 taken from Mankiw's introductory book to Microeconomics. A carbon tax and the cap-and-trade approach comes down to the issue of certainty. A tax provides for cost certainty; the cost is fixed because of the Pigovian tax. Trading permits, on the other hand, provides for environmental certainty. What's fixed is the cap itself - and it is based on an assessment of the level of emissions you need to get to in order to protect the climate. In that sense, if the cap is set too high permit prices will be low and the incentive effect will be weak. If the cap is set too low, permit prices will be very high and that can lead to the disruption of economy and trade.

2.3 Safety and Environmental Protection. At what price ?

This dissertation is mainly concerned with valuing, in monetary terms, measures that may reduce risk. This is closely related, if not equal in some cases, with measuring the actual damage that the risk poses to individuals but most importantly to the society. This Section is a short introduction on the different valuation methods. One major issue that arises in here is that the marginal damage cost is, in general, not constant; a result that is the core for the need of practicing CBA within FSA instead of CEA which is currently the norm.

As it was briefly discussed before, individuals derive utility from consuming goods and services (such as meals and holidays) and from the state of the natural environment (Hanley, 2007) and even the state of their health. In cases of regulations (or any other action) that reduce risk (or even poses an extra risk) to health and the natural ecosystem, the natural question that arises is how much risk is reduced (or increased) at what cost. So, how can we get this 'price' or 'value' of the above 'goods' ?

2.3.1 Valuation methods

Total Economic Value

The term “value” can have several different meanings (Freeman III, 2003).

“For example, economists and ecologists use the term in two different ways in discussions of environmental services and ecosystems.

Ecologists typically use the term to mean “that which is desirable or worthy of esteem for its own sake; thing or quality having intrinsic worth” (Webster’s New World Dictionary 1988). Economists use the term in a sense more akin to “a fair or proper equivalent in money, commodities, etc.” (Webster’s again), where “equivalent in money” represents the sum of money that would have an equivalent effect on the welfare or utilities of individuals.”

The unit of measurement in CBA as we discussed above is money although its conceptual basis (on welfare economics) is on utility. If CBA should be used to evaluate alternatives then all relevant cost and benefits have to be considered and valued. The benefit part is related to the benefits of preventing a change in the current state or the costs (forgone benefits) of allowing it to be degraded. We begin by saying that all goods and services must have a “value” although we recognize that sometimes it is very difficult to find the exact value or even unethical to place one. We also share the view of Epstein (2003) that a “failure to assign a dollar value to the benefits effectively assigns them a zero value or a zero weight in the calculation of net benefits, implying that changes in those services will not be incorporated into the net benefit calculation”.

As previously discussed, rational choices are based on utilities. Since there is no cardinal measure to express changes in utility, economists use money as a measure of utility in order to compare choices and to express preferences. The two money metrics used for the economic valuation of a change in price or the availability of a good or service are (Hanley and Barbier, 2009):

- (a) the most an individual is willing to pay (WTP) for that change, if this is a desirable event, or
- (b) the minimum compensation that is willing to accept (WTA) to forgo the change, if this is not desirable.

The **total economic value (TEV)** can be divided into two categories :

- (a) instrumental or use value;
- (b) intrinsic or non-use or passive value.

The instrumental value refer to those values associated with current or future (potential) use of an environmental resource by an individual, while nonuse values arise from the continued existence of the resource and are unrelated to use.

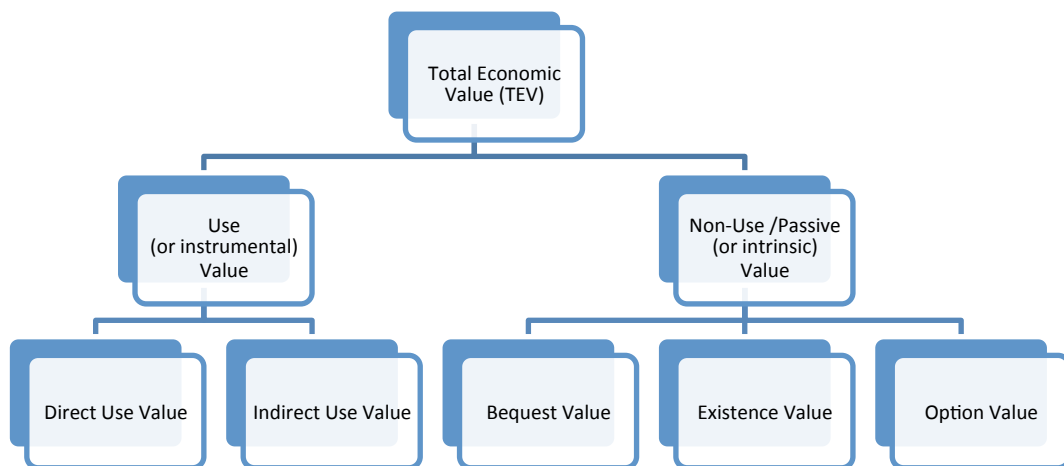


Fig. 2-8 :Total Economic Value–Adapted from Asafu-Adjaye (2005)

Use value can be further divided into direct value (which consists of consumptive values such as fishing) and indirect use value (which are of non-consumptive use, such as swimming). These values can be measured easily by using market price or other means and are well accounted for in economic appraisal.

Intrinsic (or non-use value) are inherent in the good and comprise of existence value (that is the benefit that arises from knowing that a good exists and will continue to exist regardless if it will never be seen or used), bequest value (from knowing that this good will be available for the future generations) and option value (which is related to the WTP to ensure that the resource will be available in case it is decided to be used). These non-use values, on the other hand are not traded and cannot be easily estimated (Asafu-Adjaye,2005).

Classifying Valuation Methods

Valuation methods can be separated into two broad categories of methods (Tietenberg, 2011):

- (a) Stated Preference (SP) that derive the value by using a technique that attempts to elicit the respondents' willingness to pay or willingness to accept (which is the "stated" preference)
- (b) Revealed Preference (RP) *"that are based on actual observable choices that allow resource values to be directly inferred from those choices"*.

These methods include direct and indirect techniques as presented in Table 2-1 and briefly described below, both based on Tietenberg (2011).

Table 2-1 :Economic Methods–Source: Tietenberg (2011)

Economic Methods for Measuring Environmental and Resource Values*

*Source: Modified by Tietenberg from Mitchell and Carson,1969

Methods	Revealed Preference	Stated Preference
Direct	Market Price Simulated Markets	Contingent Valuation
Indirect	Travel Cost Hedonic Property Values Hedonic Wage Values Avoidance Expenditures	Attribute-Based Models Conjoint Analysis Choice Experiments Contingent Ranking

Direct Stated Preference (SP) methods use survey techniques to elicit willingness to pay or to accept. The most popular stated preference method is the so-called Contingent Valuation (CV), the simplest version of this approach is to asks individuals what value they would place on a marginal change of the resource from its current state. On the other hand, Revealed Preference (RP) methods are "observable" because they are based on actual observable choices and can be either direct if the value corresponds to a market price (or even, a "hypothetical market" price) and "indirect" when they infer a value rather than estimate it directly.

We shall illustrate some of these methods in a hypothetical chemical oil spill (inspired from Tietenberg (2011)). It is easy to calculate how much money the local fishermen lost from by observing how much the catch declined and the resulting diminished value of it based on the market price of the fishes ("direct" calculation). Or, indirectly, we can calculate the value of an occupational health risk (such as the exposure of the same fishermen to the toxic of the oil spill by examining the differences in wages

across industries in which workers take on different levels of risk (“Hedonic Wage Method”). If for example the oil spill is near a recreational area (let’s say for sport fishery) then this damage could be estimated using the “Travel Cost Method” which uses the demand curve for willingness to pay for a visit to this area.

Finally, note that the above methods can also be used to monetize changes in human health. The use of Hedonic Wage Method was presented above. In addition, Contingent Valuation, for example, could be used to estimate the “Value of Statistical Life” by just asking individuals how much they are willing to accept for a small increase in mortality risk or willing to pay for a small decrease in that same risk. This issue is further analyzed in Section 3.7.

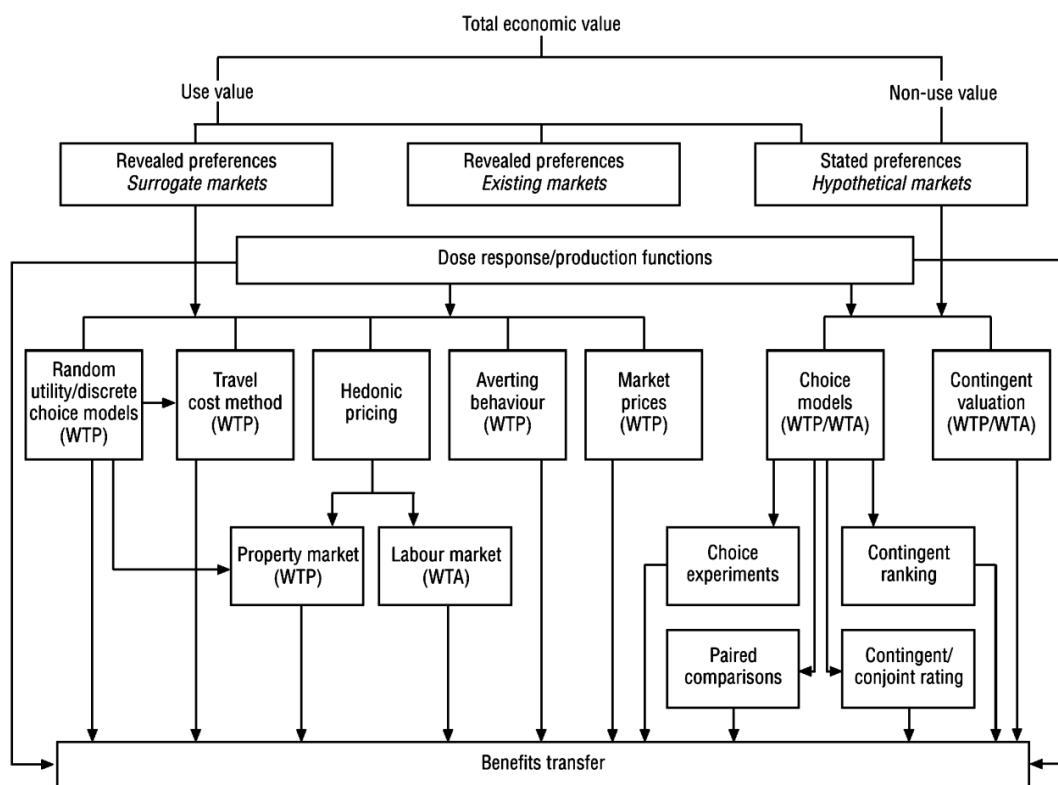


Fig. 2-9 :Value and Valuation Methods–Source: Pearce et al. (2006)

To sum up, we present Figure 2-9 that summarizes the components of the Total Economic Value and the methods that can be used to elicit these values. In addition, this Figure presents the “Benefit Transfer” (see bottom) that is a method that makes easy to use (“transfer”) values elicited (or used) in previous studies.

Noting that in this dissertation we focus on the value itself and not the techniques that can be used to estimate the value, the reader is referred to the vast amount of relevant literature, see for example Freeman (2003), Arrow et al. (1993), Carson et. al. (1996) and the handbooks edited by Foster (1997)and Maler & Vincent (2005). In

addition, the excellent book of Hanley and Barbier (2009) describes in detail and presents relevant examples of how these valuation techniques can be used in environmental-related Cost Benefit Analysis.

2.3.2 Total and Marginal Damages and Benefits and the notion of the ‘per-unit price’

Within risk analysis the major goal is to reduce risk (although this dissertation also presents the way to value increases in risk). In addition much environmental and safety policies and the related Cost Benefit Analyses involves making the current situation better; that is indeed decreasing the relevant risk. The benefits in these cases (involving risk reduction) are best thought of (measured) as damages avoided. Russell (2001, p.135) urges us to think of a situation in which pollution P causes damage $D(P)$ (in \$ per year); and if we knew the damage function $D(P)$ (with $dD/dP > 0$) and if the existing pollution level is $P_0 > P_1$, then the natural definition of the benefits of a policy (or an action) to reduce P from P_0 to P_1 (a reduction of ΔP) would be:

$$B(\Delta P) = D(P_0) - D(P_1)$$

Benefit = Damages Avoided

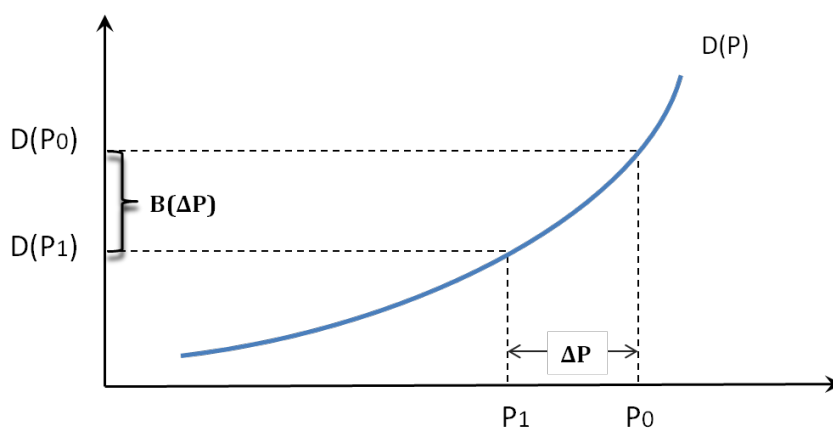


Fig. 2-10 : Typical Damage Function – adapted from Russell (2001)

Russell (2001) illustrates this graphically for a hypothetical function $D(P)$, which is the damage as a function of the amount of pollution (in physical units), and states that this observation does not solve any of the tough problems which is either obtaining the form of the total function or of the marginal function for reductions in P .

However, this observation on the shape of the damage function is useful within the concept of this dissertation for the following reason:

(a) First of all, in short, valuing risk reductions is related to the notion of willingness to pay (WTP) of the individual (or of the society as a sum of the individual WTP) that was presented above. People are indeed willing to pay for improvement in quality of the

environment and this is also true for improvements in the quality of their life, too. In a similar way, individuals (and the society) may be willing to accept (WTA) compensation for increases in the risk they face. However, note that WTA is not constrained by the income of the individual (or the budget of the society in extension) as is WTP and it should come to no surprise that when people are asked WTA questions their answers are higher than the WTP response for the same item (Field and Field, 2009).

(b) Then, according to most environmental economics (textbooks) the damage function for those “bads” that fall within the scope of this work is, in general, an increasing function, see also the Fig. 2-10 based on Russell (2001). According to Tietenberg (2011, p. 363) although the knowledge in the control and damage costs is far from complete, economists generally agree on the shapes of these relations and he continues by stating that :

“the marginal damage caused by a unit of pollution increases with the amount emitted. When small amounts of the pollutant are emitted, the incremental damage is quite small. However, when large amounts are emitted, the marginal unit can cause significantly more damage. It is not hard to understand why. Small amounts of pollution are easily diluted in the environment, and the body can tolerate small quantities of substances. However, as the amount in the atmosphere increases, dilution is less effective and the body is less tolerant.”

(c) Finally, the economic value of risk reduction equals (Hanley, 2007)

$$\text{Value of Risk Reduction} = \frac{\text{Willingness to pay for risk reduction}}{\text{change in risk}}$$

Given that the economic benefit from reducing risk (ΔB), which is equal to the reduced damage, can be approximated by the willingness to pay for risk reduction it is obvious that the per unit value of risk reduction equals the marginal damage cost.

In most cases, the damage function is not proportional to the quantity (as discussed in the previous point) and, therefore, there is no evidence to assume that the marginal cost and, in extension, the per-unit value of risk reduction is, in general, constant.

This is a major result, which can be also extended for risk reductions (or increases) that affect the quality of human lives. This comes to a surprise to those that are familiar with Formal Safety Assessment since in most cases health effects are assessed by the so-called *Implied Cost of Averting a Fatality* (ICAF) which is related with the willingness to pay for risk reductions. This is also related to the so-called Value of Statistical Life (VSL or VOSL) that has been extensively used in Cost Benefit Analysis. To start with, it may again come to a surprise to those that are **not** familiar with Formal Safety Analysis that many experts involved in FSA studies do strongly oppose the use of this value within FSA. This issue is further discussed in Section 3.7.

On top of that, most experts involved in FSA studies do probably accept a constant ICAF value (although the term itself is rarely being used nowadays) and do also believe that a constant unit price is the case when assessing risks other than safety, see for example case with oil pollution. Something that is obviously not valid as we described above.

Going back to the constant unit cost, why do experts use a constant willingness to pay as the value of statistical life? Is that correct? In what effects may this assumption be valid? Well, although it can be debated, a constant marginal willingness to pay to avert the risk of a statistical fatality (i.e not proportional to the risk reduction) is probably correct. The reason is that in quite all cases we do assume small risk reductions. There are indeed two cases where the per-unit damage cost (or cost to avert) can be assumed constant: (a) when the reduction is very small and (b) when the damage cost is truly proportionate to the quantity.

In the first case, a constant unit cost is indeed a good approximation for all risk reductions (even if we are talking about pollution or any other kind of risk) as long as the risk reduction or the related background risks are small. As you may notice in Fig 2-x, the unit price is the marginal damage, which is the slope. This is nothing new; Hammitt of Harvard University has written about that over and over. *“Economic theory suggests that WTP should increase almost proportionately to the size of the risk change”* (Robinson and Hammitt, 2010). Empirical studies have also found that individuals state a constant WTP for small risk reductions and one possible explanation may be *“that individuals are indifferent between risk changes of these magnitudes”*. (Robinson and Hammitt, 2010). Note that in the case of human safety risk reductions are small. In our opinion, the non linearity would be obvious if the amount of loss of lives averted were quite large.

The second case is not so common. However, some papers on climate change damage estimation have advocated the use of uniform per-unit values for damages (e.g. Hohmeyer and Gaertner, 1992; Meyer and Cooper 1995). Although this is debatable, some Environmental Economics textbooks support this may be indeed true for the so-called *stock pollutants*. According Tietenberg (2011) damage done by the amount of wastes emitted depends on the capacity of the environment to assimilate the waste products, which is the so-called *absorptive capacity of the environment*. Pollutant for which the environment has little absorptive capacity (such as most waterborne toxic pollutant, inorganic chemicals like plastic, heavy metals, synthetic chemicals such as dioxin) are called *stock pollutant*. In the case of stock pollutants damage caused by their presence increases but is assumed to be proportional to the size of the accumulated stock and persists as the pollutant accumulates. Although these wastes

are out of the scope of this work these may be one case where the damage is proportional to the quantity emitted and, thus, the marginal cost (or per-unit price) can be assumed constant.

However, this is not the case for *fund pollutants* (including contamination of water from oil spills and all major ship air emissions) for which the environment has some absorptive capacity. Note, that although carbon dioxide (CO₂) accumulates in the atmosphere (and that is the reason for being associated with Climate Change) it is absorbed by plant life and the oceans and therefore is categorized as a fund pollutant.

2.4 Introduction to Cost Benefit Analysis

CBA is an accounting technique for capturing the advantages and disadvantages of an action in monetary terms, see Krupnick (2004). This action can be a project, a Risk Control Option (RCO), a medical intervention, a policy or any other measure. Subtracting costs from benefits yields the net benefits to society (also referred to as net improvements in social welfare). Actions that improve welfare or well-being are superior to those that reduce it. Furthermore, CBA can be used to cardinally rank them on the basis of their change in well-being. CBA focuses on the aggregate measures of well-being, taking the existing distribution of income as given.

The basic criterion is that if the discounted present value of the benefits exceeds the discounted present value of the costs then the action is worthwhile. This is equivalent to the saying that the net benefit must be positive or that the ratio of the present value of the benefits to the present value of the costs must be greater than one. Theoretically speaking, the higher the ratio the better the regulation is. However note, that the ratio criterion should be avoided, see Section 2.5.6. The above-mentioned equivalent criteria are the following:

$$B > C \quad \text{or} \quad B - C > 0 \quad \text{or} \quad \frac{B}{C} > 1$$

In general, the **cost** component consists of the one-time (initial) and running costs of an RCO, cumulating over the lifetime of the system. The **benefit** part is much more intricate. It can be a reduction in fatalities or a benefit to the environment or an economic benefit from preventing a total ship loss. Cost is usually expressed using monetary units. To be able to use a common denominator, a monetary value has to be given for the benefit too.

CBA is indeed a complex process and has practical difficulties in each of its steps. The major steps are more or less well accepted and are the following (Boardman et al., 2001):

1. Specifying the set of alternative projects
2. Decide whose benefits and costs are standing
3. Catalogue the impacts and select measurement indicators (units)
4. Predict the impacts quantitatively over the life of the project
5. Monetize all impacts
6. Discount costs and benefits to obtain present values
7. Compute net present values (NPV) for all alternatives
8. Perform sensitivity analysis
9. Make recommendations based on Step 7 and 8.

CBA starts with the definition of the scope and the specification of alternatives. In the scope of this work these alternatives are the risk control measures that are defined in the previous steps of the risk assessment. Then, the population (or stakeholders) that stand the costs and the consequences are decided. In the next step, the analyst catalogues the impacts that affect the utility of individuals with standing and predicts them over the life of the project. Note that in shipping the life time of a ship is in the range of 20-30 years. Sometimes we may choose to assess the impacts for shorter time periods. On the other hand, there are really hard cases like ship air emissions where the impacts of today's action will be visible in the far future.

Next step, and the most controversial one, is to monetize all impacts. Monetizing means assigning value in a monetary unit, for example US dollars. It should be noted that this values, or 'prices', do not have the same meaning as the price of a product. These values are more or less used because there is the need to quantify the impacts in a common unit in order to perform calculations.

Then benefits and costs should be discounted to obtain present values and the difference between the present value of the benefits and the costs, which is the net present value, is calculated for each alternative. Given the high uncertainties involved in the prediction of impacts it is advisable to perform a sensitivity analysis to investigate the effect of the most important parameters to the net present values. Finally, the alternatives that have the largest NPV, also taking into the account the results of the sensitivity analysis, should be recommended for implementation.

2.4.1 Key Issues within Cost Benefit Analysis

Although CBA may appear at a first glance as something that can be easily performed there are many points where the analyst should be cautious. Below some key issues will be briefly analyzed so that the reader can follow the discussion in the next Sections. For more details the reader is referred to CBA textbooks for example Mishan and Quah (2007), Boardman et al. (2001), de Rus (2010) and Hanley and Barbier (2009), the latter focusing on CBA and environmental policy.

1. The rationale behind CBA is based on welfare economics

As discussed in Section 2.1, CBA has a very strong theoretical background based on welfare economics and its rationale is based on what is known as the Kaldor-Hicks criterion: *“A policy should be adopted if and only if those who will gain could fully compensate those who will lose and still be better off”*(Boardman et al., 2001). In theory there needs to be no compensation, the compensation is hypothetical. However if losers are compensated this will result in a Pareto improvement. Therefore, Kaldor-Hicks criterion provides the basis for the CBA rule which demands that the regulator adopts those policies that have positive net benefits. (de Rus, 2010).

2. The importance of discounting

Benefits and costs occurring in different time periods within the lifetime of the project have to be aggregated to obtain the net present value (NPV). However, for most individuals a dollar today has more value than a dollar after one year. Actually, assuming a constant discount rate ($0 < r < 1$) a present amount P has a value of F at some future point in time after T periods (usually years) can be estimated by the following simple equation :

$$F = P \cdot (1 + r)^T \quad \text{Eq. 2-1}$$

All actions have an associated flow of costs and benefits during their life time (T years) that have to be added to obtain the NPV. The net present value (NPV) of implementing a risk control measure is calculated using the following equation:

$$NPV = \sum_{t=0}^T \frac{B_t - C_t}{(1 + r)^t} \quad \text{Eq. 2-2}$$

where B_t are benefits in period t ;

C_t the costs in period t ;

r is rate used for discounting (per period); and

T the number of periods (usually years) the project will last.

To discount a flow of n equal amounts A (can be costs or benefits) that incur at regular intervals (f.e at the end of each period for a total of T periods) assuming that the discount rate is constant, then it can be easily shown (see for example Brealey and Meyers (2003) and de Ruis (2010)) that the present value P of for this money flow is:

$$PV = A \cdot \left[\frac{1}{r} - \frac{1}{r(1+r)^T} \right] = A \cdot \left[\frac{(1+r)^T - 1}{r(1+r)^T} \right] = \sum_{t=1}^T \frac{A}{(1+r)^t} \quad \text{Eq. 2-3}$$

Obviously the choice of a discount rate can be difficult, but many organizations that conduct analyses have standard rates to use in their analyses. For example, the UK Health and Safety Executive (HSE) uses an effective discount rate for health and safety benefits of 1.5% for benefits accruing up to 30 years in the future. Furthermore, US Office of Management and Budget in Circular A-94 (Guidelines and Discount Rates for Benefit-Cost Analysis of Federal Programs) gives an upper bound of 7% for federal projects. In most FSA studies values of 3% and 5% have been used extensively. Note that hopefully enough lifetime of risk control measures regarding vessels is maximum 20 to 30 years.

However, discounting over very long time horizons is complicated mainly because these investments involve greater uncertainty and in some cases (for example, regarding CO₂ future generations are affected. For this reason, ethical arguments for using low or zero discount rates for long-term projects are widespread. One of the most accepted practices is the use of hyperbolic discounting which is to raise the initial discount rate relative to the exponential rate and then lower the rate in later years. This is to overcome the concern of environmental economists that believe using exponential discounting (which is what we do in projects with short lifetime) penalizes risk control option (or projects) whose benefits are realized in the long term (f.e reducing air emissions). Unlike exponential discounting which is what we do in projects with short lifetime, with hyperbolic discounting the present value of benefits and costs that occur in the distant future does affect the profitability of the project (de Ruis, 2010).

3. Selection between actions with different lifespans

Obviously, things are easy when a CBA is being conducted for the economic appraisal of a single project however that is not the case when choosing among alternatives. Alternatives must be comparable which actually goes beyond having the same discount rate. Alternatives must have the same lifespan. One way to overcome this difficulty is to calculate the so-called equivalent annual net benefits or cost, see for example Boardman et al.(2001), Brealey and Meyers (2003) and de Ruis (2010).

4. Decision Criteria

In the literature, a couple of criteria for economic appraisal are being used, for example those based on the net present value, the cost benefit ratio and the internal rate of return (the interest rate for which NPV=0) to name a few.

In the case of only one action (this can also be seen as an alternative to the 'status quo') the decision rule is simple : *An option (action, project or risk control option) that has a positive NPV should be considered to be recommended for implementation.* Both the criteria based on ratios (cost/benefit or benefit/cost) and the NPV criterion will yield the same result. However choosing among alternatives is much more complicated. Almost all textbooks on CBA and many on finance agree that NPV should be the only criterion used. In Brealey and Meyers (2003) there is a whole chapter entitled '*Why Net Present Value Leads to Better Investment Decisions Than Other Criteria*'. One of the main reasons is that ratios do not take into account the scale, see Section 2.5.6.

2.5 Introduction to Cost Effectiveness Analysis (CEA)

CEA, which is the most common method used in health economics (see Garber and Phelps, 1997; Weinstein and Manning, 1997;Gold et al., 1996;Brouwer and Koopmanschap, 2000) is a considered by many (e.g. Mishan and Quah,2007; Krupnick,2004) as particular form of CBA, where the benefits are usually not monetized, and therefore, net benefits cannot be calculated. In the case of "net cost-effectiveness analysis" any monetized benefits of a policy are subtracted from costs. Usually, in CEA, one calculates costs per unit of an effectiveness measure (such as lives saved). Therefore, while CEA cannot help in determining whether a policy increases social welfare, it can help in the choice of policy that achieves the specified goal with the smallest loss in social well-being and can help rank alternative policies according to their cost-effectiveness (Krupnick, 2004).As briefly discussed in Section many consider CEA to have it theoretical background in extra-welfarism (e.g. Culyer, 1991)

2.5.1 Incremental cost-effectiveness ratios (ICER)

The analytical tool of CEA is the **incremental cost-effectiveness ratio (ICER)**, also called marginal cost-effectiveness ratio, given by the difference in costs between two actions divided by the difference in outcomes between these two with the comparison typically being between an action that is proposed to be implemented and the current situation. The use of CEA and ICER is extensive in health economics where the ICER value aids the decision maker to adopt or not health-related programs.

In the current FSA guidelines the cost to avert criterion that we presented above uses a similar approach. To be more specific, in the context of FSA, an incremental cost-effectiveness ratio is used but in most cases benefits are also taken into account (that is the case of net CEA), however, benefits are also monetized. Similar, cost-effectiveness indices can be constructed to assess environmental risks, such as oil pollution and GHG emissions. In the scope of this paper, the following indices can be formulated as a generalization of the CAF index :

Gross Cost Effectiveness Index (GCEI)

$$GCEI = \frac{\Delta C}{\Delta R} \quad \text{Eq. 2-4}$$

Net Cost Effectiveness Index(NCEI)

$$NCEI = \frac{\Delta C - \Delta B}{\Delta R} \quad \text{Eq. 2-5}$$

where

ΔC is the cost per ship of the action (eg. measure, risk control option) under consideration (\$)

ΔB is the economic benefit per ship resulting from the implementation (\$), and

ΔR is the risk reduction per ship, in terms of the number of lives averted.

Note that in the FSA Guidelines ΔR is the risk reduction in terms of fatalities averted and the relative indices are called **NCAF** (Net Cost of Averting A Fatality) and **GCAF** (Gross Cost of Averting a Fatality). The rationale of both indices can be extend to other, than health and safety, effects such as pollution, as it will be analyzed in the next 3 Chapters. The net index can also take into account such monetary risk in the ΔB . However, the index can only use one non-monetary effect, that will be ΔR .

In addition, from a theoretic perspective, when the two actions are independent –that is, where costs and effects of one action are not affected by the introduction of the other action – and those that are mutually exclusive– that is, where implementing one measure the other cannot be implemented- then the cost effectiveness ratios can also be used. In that sense, the differences used in the above ratios are also misleading since we are talking about actual costs and not differences.

In general, the **cost** component consists of the one-time (initial) and running costs of an RCO, cumulating over the lifetime of the system. The **benefit** part is much more intricate. It can be a reduction in fatalities or a benefit to the environment, as explained further below, or an economic benefit from preventing a total ship loss. Cost is usually expressed using monetary units. To be able to use a common denominator, a monetary value has to be given for the benefit too. However, the outcome could also be measured in units relevant to the purpose of the assessment, that is, tonnes of oil

spilt averted in the case of oil pollution, tonnes of carbon dioxide averted in the case of CO2 emissions and so on (see relevant Sections).

After the estimations on cost and benefit, these values are combined with the **effect** of the measure (the so-called 'Risk Reduction' in the FSA context) by using the cost effectiveness indices. Change in outcomes can be negative or positive; however we expect that an FSA should aim to reduce the harmful outcome. Note that the case of negative changes in outcomes is an interesting one and deserves further investigation.

2.5.2 ICER Thresholds

In the healthcare sector, programs with ICERs that lie below a specific threshold, referred to as λ value, are deemed to be cost effective and should be adopted. In most of the cases, λ -value is expressed in \$ per QALY, that is Quality Adjusted Life Year (see Section 3.7 below). According to the Dictionary of Health Economics (Culyer, 2008) the ICER threshold is *"the maximum acceptable incremental cost-effectiveness ratio acceptable to a decision maker. Beyond this threshold, health care technologies will not be adopted on efficiency grounds alone."*

Generally speaking, a threshold λ is a value expressed in monetary terms or units such as US dollars per number of fatalities averted, US dollars per tonne of pollutant (CO2, oil etc) averted and so on.

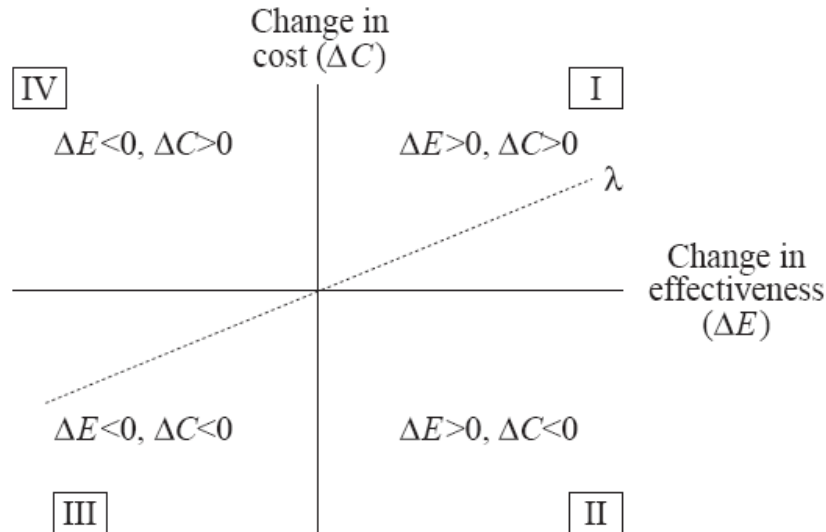


Fig. 2-11: The cost effectiveness plane - Source Culyer(2005)

Fig. 2-11 illustrates the four-quadrant figure of cost difference plotted against effect difference. Actions that lie in II and IV quadrant have negative net cost effectiveness (CEI) ratio and should be addressed with extreme caution. Let there be a maximum amount a decision maker will pay for an increment of outcome (ΔE) indicated by the dotted line λ . All points below λ are in the 'region of acceptability' (note that $\Delta C/\Delta E$ is

lower – actually negative – than λ in quadrant IV, but that this quadrant cannot be in the region of acceptability since ΔE is negative).

We want to elaborate a little on the important case of negative cost effectiveness indices especially due to an increase in the outcome (ΔE is negative). Risk Analysis can also be used to assess such cases, although by definition this is not the case when using Formal Safety Assessment. Note that in this case an action is cost effective if, and only if the ICER is GREATER than λ (see the Cost Effectiveness Plane illustrated in Fig.2-11). That is the reasoning behind the need to use the absolute values of $(\Delta C - \Delta B)$ when dealing with negative NCEI.

Furthermore, Fig.2-12 was constructed under the standard assumption that there is a little difference between a person’s willingness to pay (WTP) and willingness to accept (WTA), see O'Brien et al. (2002) and Klok and Postma (2004). The WTA is the compensation that a person is willing to accept in the case of negative outcome. According to basic utility theory (see Kahneman and Tversky (1979)) WTA should be greater than WTP. Furthermore, they argue that this disparity is a result of the so-called “endowment effect” according to which people value goods more highly once they own them – perhaps the result of a sort of loss aversion whereby choices are seen in terms of gains or losses relative to their current endowment. Other reasons for this disparity can be found in O'Brien et al. (2002) and Klok and Postma (2004).

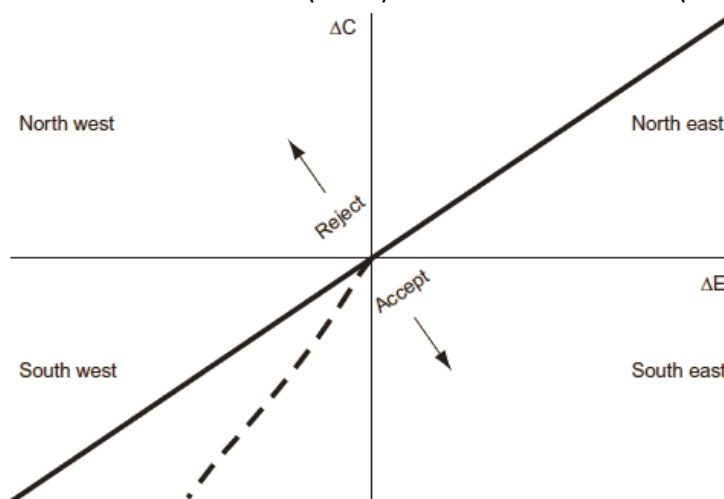


Fig. 2-12: The cost effectiveness plane – Source: Klok and Postma (2004)

Thus, there is enough evidence that the threshold λ that divides the plane into the cost effective and non cost effective regions is NOT a straight line. This means that in the cases where the ΔE is negative the effectiveness threshold should be different. In the context of the present work negative change in outcomes will not be further analyzed but is definitely an important area for future research.

2.5.3 More on the decision rules of CEA

Johannesson (1994) has stressed out the misunderstood distinction between incremental (marginal) cost effectiveness ratios and average cost-effectiveness ratios which is important *"in order to differentiate between choices among independent programs (e.g., blood pressure control versus ulcer treatment) and choices among mutually exclusive programs (e.g., different drugs to control high blood pressure)."*

The decision rules based on Weinstein and Zeckhauser (1973) and Weinstein (1990) that were described in Johannesson (1994) shall now be presented. The optimal decision rule in order to choose between a number of independent programs (under a budget constrain) is to rank the programs from the lowest to the highest cost effectiveness ratio (e.g., dollars per averted life) and to select programs from this ranking list until the budget is exhausted (Weinstein and Zeckhauser, 1973).

On the other hand, there are cases where a number of different alternatives (mutually exclusive options) are available. In this case, the alternatives should be first ranked according to their effectiveness (in terms of risk reduction) and then the incremental cost-effectiveness ratio for each successively more effective program should be calculated (e.g., the incremental cost divided by the incremental gain in effects). If any of these incremental ratios is less than the previous one in the ranking of increasingly more effective (mutually exclusive) programs, then the less effective one is excluded from the list as dominated, and it should never be implemented. This is irrespective of the size of the budget (Weinstein,1990). Then, the incremental ratios should be calculated again (with the dominated alternative excluded), and this process is repeated until a number of programs with increasing incremental cost-effectiveness ratios remains. This algorithm results in an ordering of programs (with increasing incremental ratios) and the optimal decision is to move up the list of incremental ratios and successively replace a less effective program with a more effective program until the budget limit is reached. In the case of mutually exclusive programs we are not talking about ranking lists since only one program can be implemented and that will be the one that meets the budget constrain. Note that a similar procedure can be followed in the generic case where a number of clusters of mutually exclusive programs are available.

2.5.4 Priority of Indices

It can be seen that if $\Delta B > 0$ (a reasonable assumption if the measure in question will result to some positive economic benefit), then if the measure satisfies the GCEI criterion ($\Delta C < \lambda \Delta R$), it will always satisfy the NCEI criterion as well ($\Delta C < \lambda \Delta R + \Delta B$). In that sense, the GCEI criterion dominates the NCEI one. The opposite is not necessarily the

case. Similarly, as a result of this property, it has been proposed by many FSA reviewers that first priority should be given to GCAF, as opposed to NCAF (see the FSA Guidelines).

2.5.5 Negative Cost Effectiveness Values

When comparing two actions a negative ΔE value may be reasonable. However, in most cases a risk assessment is performed in order to implement an action which will increase the current safety level or reduce the risk of environmental harm which implies a positive ΔE . Thus, when figures of GCEI and NCEI are positive, their meanings are understandable. However, when the values become negative this may be more difficult. Safety measures come usually at a cost and, thus, we expect a positive ΔC . On the other hand, there are cases, where cost may be negative. Take, for example, the case of energy-efficient lamps. These may be more expensive than normal light bulbs but their use lead to money savings (negative costs) and a decrease in CO₂ emissions. Thus decision makers should be very cautious with negative net CEI.

A negative NCEI (see Eq.2-4) means that the benefits in monetary units are higher than the costs associated with the measure. As proposed in document MSC 76/12, within FSA studies, when comparing RCOs whose figures of NCAF are negative, the absolute values of $(\Delta C - \Delta B)$ should be used.

Furthermore, there is always the possibility of extreme negative values as a result of extreme higher benefits than costs and small reduction in the effects. This is in line, with the consolidated FSA Guidelines (MSC 83/Inf.2) which states that *“RCOs with high negative NCAFs should always be considered in connection with the associated risk reduction capability”*.

Apart from benefits overestimation, negative values are important to identify market imperfections. This issue has attracted much attention lately regarding emission reduction measures and is often referred to as the “energy efficiency gap”. In a perfect market firms or society would adopt any measure that has a negative net cost. There are indeed some “low hanging fruits” especially those that have large negative costs and come with large emissions abatement potentials.

High negative values can be very valuable in identifying imperfections such as lack of information which is usually an important reason for not implementing cost effective measures. Other imperfections are related to the so-called ‘split incentives’. For example, some firms may be unwilling to be extremely costly protection measures for workers since it is the company that pays the cost but the workers are those that are benefit from this.

2.5.6 Beware of the ratios

Ratios do not take into account the different scales of projects. Scale is not a problem if the level of effectiveness (or the cost) is constant across all alternatives, which is not the case in general.

Cost and Effectiveness	ALTERNATIVES	
	A	B
Cost Measure (budget cost)	\$1M	\$100M
Effectiveness Measure (number of lives saved)	4	200
CE Ratio (cost per life saved)	\$250,000*	\$500,000
EC Ratio (lives saved per million dollars)	4.0 lives*	2.0 lives

*CE ratio or EC ratio of the most cost-effective alternative

Fig. 2-13: The cost effectiveness plane – Source: Boardman et al. (2008)

See for example, the figure above which is taken from the excellent CBA textbook of Boardman (2008). Clearly, if we use the cost-effectiveness ratio as the decision rule then we would choose alternative A. Yet, if we look more closely at alternative B, we see that it would save a large number of lives at the relatively low value per life averted of \$0.5 million which is even well below the \$3 million used in FSA any may other WTP values found in the literature.

Both NCAF and GCAF that are extensively used in FSA suffer from the 'scale effect'. The following example is taken from Kontovas and Psaraftis (2009). Fig 2-14 presents 2 alternatives. In this case, both RCOs are acceptable, since both have GCAF and NCAF below \$3m. Also, RCO2 is superior to RCO1 in terms of both criteria (having lower ratio). However, RCO1 reduces fatality risk 10 times more than RCO2, meaning that in this case the RCO that is selected as best is expected to reduce risk 10 times less than the one that is rejected! Now, assume that another alternative, RCO 3, is available. This RCO has the same gross cost effectiveness ratio with RCO 2 but a greater net cost effectiveness ratio. In this case it is difficult to choose among these two RCOs but it is certain that these alternatives are not equal.

	ΔR	$\Delta C (\$m)$	$\Delta B (\$m)$	GCAF (\$m)	NCAF (\$m)
RCO 1	0,1	0,1	0,09	1	0,1
RCO 2	0,01	0,009	0,0085	0,9	0,05
RCO 3	0,1	0,09	0,0085	0,9	0,815

Fig. 2-14: Hypothetical example leading to selection of most risky RCO
Source: Kontovas and Psaraftis (2009)

To explain the paradox, we note again that being ratio tests, both GCAF and NCAF ignore the absolute value (or scale) of risk reduction DR, which should always be taken into account as criterion in itself. If anything, comparisons should be made among alternatives that have comparable DR.

2.6 The link between CEA and CBA

A key issue is whether there is a link between CEA and CBA. This issue has been addressed times and again in Health Economics and according to Johannesson (1995) with a constant willingness to pay per unit is used to estimate the benefit, CEA and CBA will yield the same results. Although there are divergent opinions on the subject it is obvious that a cost effectiveness analysis that takes into account all the costs no matter who bears them and without having any budget limit then CBA and CEA do arrive at the same results. In order this to be valid the benefit of the RCO (which in our case is equal to the damage avoided by introducing the measure) should be equal to the quantity $\lambda\Delta R$, where the cutoff value λ is the constant per unit cost for a change of ΔR . That can be easily illustrated in the following two criteria.

$$\text{Cost Effectiveness Analysis: } GCEI = \frac{\Delta C}{\Delta R} < \lambda \Rightarrow \Delta C < \lambda \cdot \Delta R \Rightarrow \Delta C - \lambda \cdot \Delta R < 0$$

$$\text{Cost Benefit Analysis: } \Delta C < \Delta B = \lambda \cdot \Delta R \Rightarrow \Delta C - \lambda \cdot \Delta R < 0$$

Phelps and Muslin (1991) argue that the major difference between these two methods is that CBA states the dollar value of the outcome (for example dollar value of life or QALY) explicitly whereas CEA does so implicitly as the two equations above illustrate. Furthermore they state that CEA estimates “prices” of the outcome and that the analysis asks if the price is acceptable and, on the other hand, decisions based on CBA depend upon the net value. Moreover, Bala et al. (2002) in line with the above argue that the distinction between them is that “*the objective of the CEA is to determine the least costly way to achieve a goal, while the objective of CBA is to determine whether the goal is worth achieving*”. Dolan and Edlin (2002) considered the attempts to link CBA and CEA by Johannesson and Meltzer (1998) and Bleichrodt and Quiggin (1999) and argue that it seems that the restrictions in order to arrive at an equivalency are unrealistic.

However, within the scope of this work it is noted that the above can bridge CEA and CBA in the case of constant returns to scale. This is probably true for small risk reductions and mainly when assessing effects on health and safety. However for environmental projects, given that most damage functions are non-linear (see section 2.3.2) there is probably no bridge between these two methods; this is supported also by other researchers. For instance, Gafni (2006) notices the recent growing interest in the use of CEA (using Quality Adjusted Life Year (QALY) as an alternative to WTP) as the method of evaluation for environmental projects and concludes that there is neither theoretical nor practical support for the claim about the superiority of CEA.

2.7 On the inconsistency of using constant CEA decision rules

This Section deals with the rather important issue of constant thresholds, both for net and gross cost effectiveness criteria. Currently, following the constant 'price' approach, the alternative with the highest incremental cost-effectiveness ratio that is equal to or below the stipulated willingness to pay for one unit of effectiveness should be used (Karlsson and Johannesson, 1996). To that extent, many FSA analysts, as it will be discussed in the next Chapters, are advocating the use of constant thresholds to address the cost effectiveness of measures that reduce oil spills and ship air emissions. This tendency is often based on a misunderstanding of the underlying theory and probably the extreme visibility that constant thresholds were gained after the use of such a criterion for the case of measures that reduce risk to human life, which is also included in the FSA Guidelines. The reader is referred to Section 4.2 for the constant threshold for oil spills which was initially presented by Skjong et al.(2005) and to Section 5.4 for a net cost effectiveness index to address measures that reduce emissions that was proposed by Eide et al. (2009).

We start our critique by noting that the use of a constant threshold and thus of a constant marginal utility of health effects has been also questioned lately in health economics; an area where the use of constant thresholds was predominately. However, most of the critique is performed from a different aspect of view, that of the returns to scale. In short a constant return implies that the costs and effectiveness per measure is identical for all patients. For the critique, see Birch and Gafni (1992),Sendi et al.(2002), Donaldson et al (2003). Elbasha and Messonnier (2003) question the constant thresholds and more specifically the decision rules under constant returns to scale and propose an alternative method of health care resource allocation.

In the case of maritime risk assessment, as discussed in the previous Section the CEA ratios that are presented, especially the ones based on the net cost, are equal to the CBA criteria. An project (or policy) is acceptable when the net present value is greater than zero which is equal to demanding that the net cost effectiveness is less than λ ; where λ is the stipulated willingness to pay (or accept) for one unit of effectiveness as discussed in the previous section. However, in most cases there is no constant per unit cost for a change of ΔR (Section 2.5.3)! Thus in most cases, a constant threshold is meaningless. Some exceptions include the cases of small risk reductions and when there is indeed a constant 'price'.

2.8 Further critique and some advocacy on the use of CBA

Firstly, note that CBA has a strong foundation based on welfare economics, see the Kaldor-Hicks criterion. On the other hand, it seems that cost-effectiveness analysis, in general and as it is applied within FSA, has some strong limitations. In theory, CEA is based on the notion of extra welfarism. Furthermore, the net Cost Effectiveness Index can be easily miscalculated as benefits are subtracted from cost and there is some kind of double counting since the quantity $\lambda\Delta R$ may also include benefits from the implementation of the RCO. In any case, if an RCO produces gains in more than one dimension the best way to make them consummate is by using CBA that can convert each type of benefit into a common metric, see Phelps and Muslin (1991).

In extension, a very difficult task is to take into account all relevant impacts. Furthermore, probably the main disadvantage of CBA is that it seeks monetization of all the effects. Some people feel that it is unethical to place a monetary value on health or mortality risk changes because it seems that CBA places a value on human life, see Krupnick (2004). Without entering into further detail, this view reflects a misunderstanding about the valuation process. This is due to the way that the society perceives risk. An oil spill harms the environment sometimes by killing birds and creates economic damages in the local society f.e in fishing and in tourism. On the other hand, people are not that much aware of the damage of air emissions, e.g. greenhouse gases since the effects in the climate change will take many years to appear and presumably place lower monetary value in the latter case and may avoid to value impacts of emissions.

On the other hand, one may also feel uneasy about selecting alternatives on the basis of ratios as they ignore the different scale of ratios. Scale is not a problem when all of the policy alternatives have the same cost or when the level of effectiveness is constant for all alternatives. This is not usually the case and the effect of scaling is not a problem since CEA can be modified to deal with scale differences. According to Boardman et al. (2001) there are a couple of ways, for example to select the project with the lowest cost that meets a certain effectiveness constraint or the project that yields the largest number of units of effectiveness subject to a budget constraint.

Furthermore, CEA besides being easy to be performed (actually it is cheaper and quicker) has as discussed in Section 2.1 *“the objective of the CEA is to determine the least costly way to achieve a goal”*. This has to do with the fact that CEA has to do with budget constrain. The measures investigated in a CEA can be ranked by their cost-effectiveness and the most preferable option is the one with the lowest

cost-effectiveness ratio (least cost per unit of effectiveness). If the available budget is known), it is possible to rank the measures based on how much of the desired effect can be achieved with the given budget. See Section 2.2.5 for the decision rules applied in CEA.

The role of CEA, as a methodology to help decision-makers maximize health gains from the use of limited resources, has not been changed over the years (Gold et al., 1996). Weinstein and Stason (1977) argue that *"...the underlying premise of CEA in health problems is that for every given level of resources available, society (or the decision-making jurisdiction involved), wishes to maximize the total aggregate health benefit conferred"*. This is seen as an advantage of CEA over CBA *"as the question of how to maximize health improvements generated by a given level of resources has an obvious attraction for governments"* (Gafni, 2006).

However, in the same sense, shortage of funds forces private companies to choose between worthwhile projects. Economists call this capital rationing and there are methods that can be used to select a set of projects that is within the company's resources but also gives the highest possible net present value. Commenting on these methods is out of the scope of this work and the reader is referred to Brealey and Meyers (2003). Budget constraint is not a problem when applying CBA.

However, the most important reason for which CBA is preferred in our case it that CEA inevitably omits impacts that would be included in CBA. Indeed, CEA typically considers only one measure of effectiveness. According to Boardman et al. (2001) one way to get closer to doing this-that is, to reach what they call a "halfway house" between standard CEA and CBA by computing the following ratio:

$$\tilde{CE} = \frac{\text{social costs} - \text{other social benefits}}{\text{effectiveness}} \quad \text{Eq. 2-7}$$

In any case, Boardman et al. (2001) argue that *"most likely CEA was selected in the first place because some social costs and benefits could not be monetized"* and conclude that *"for these reasons, moving all the way to CBA with extensive sensitivity analysis is often a better analytical strategy overall than expanding the scope of measured costs in CEA."*

In the case of Formal Safety Assessment what is being performed is actually a cost-effectiveness analysis which will be discussed in the next section. Obviously there are quite many questions on the use of CBA and CEA, some of which are justified. However, since CBA is a very important step within risk assessment it is advised to

performed steps that are usually missing, such as the sensitivity analysis step. This way risk assessment will become more robust, see Chapter 7 for more.

As a conclusion, and it must be clear by now, Cost Benefit Analysis, besides its strong theoretical basis on welfare economics, is the most reliable way to consider actions with multiple effects. Noting that much research has to be done to arrive at a commonly accepted non linear function (especially for cases with large risk reductions), the reader is referred to Chapter 6 for the theory and examples of how non-linear function can be used within Cost Benefit Analysis and Risk Assessment.

3

THE EFFECT OF RISK ON HUMAN HEALTH

3.1 Introduction to Human Losses due to Shipping

Inevitably all human activities pose some risk to human life. Safety is one of the major goals of all industries. Shipping poses some risk to passengers and to workers that voluntarily take the risk. Safety has been an issue for the IMO for a long time (see the SOLAS convention). Lately, IMO has introduced FSA as a way to assess risk to human life taking into account the work that has been done regarding occupational safety mostly in the UK. There is a vast literature in this subject which is out of the scope of the present report. Formal Safety Assessment has been dealing with safety since its introduction. The so-called 'Cost of Averting a Fatality' has been the major cost effectiveness index in all FSA studies and will be discussed in the next Section. Furthermore, some updated analysis of the relative threshold will be presented and as well as a discussion of the potential non linearities that are involved in the valuation of measures that reduce the risk to human life.

3.2 Cost Effectiveness Index and The 'Cost of Averting A Fatality'

Currently, only one such index is being extensively used in FSA applications and that is the one to assess risk on human safety. This is the so-called "Cost of Averting a Fatality" (CAF) and is expressed in two forms: Gross (GCAF) and Net (NCAF). These two indexes are the incremental cost-effectiveness ratios (in gross and net form) for risk reductions in terms of the number of fatalities averted, in line with what is presented in Section 2.5.1. In all recent Formal Safety Assessment (FSA) studies, cost effectiveness is limited to measuring fatality risk reduction using the \$3m criterion. This criterion is to cover fatalities from accidents and implicitly, also, injuries and/or ill health from them. There are two other criteria that were submitted at the same time with the above-mentioned criterion to the IMO but were never used. One is to cover only risk of fatality and another to cover risk from injuries and ill health. Both have a value of \$1.5m. However, we know of no FSA that has used these criteria.

More specifically the GCAF index is the gross incremental cost effectiveness ratio and is defined according to the FSA Guidelines as follows:

Gross Cost of Averting a Fatality (GCAF)

$$GCAF = \frac{\Delta C}{\Delta R} \qquad \text{Eq. 3-1}$$

where

ΔC is the cost per ship of the RCO under consideration.

ΔB is the economic benefit per ship resulting from the implementation of the RCO.

ΔR is the risk reduction per ship, in terms of the number of fatalities averted, implied by the RCO.

Note that the NCAF index is similar to the above one with the difference that in this case benefits are also subtracted from the costs.

For the GCAF criterion, the equivalent inequality is simpler: $\Delta C < \$3m \cdot \Delta R$

Note that according to Section 3.3, if the price of \$3 million is used as the constant willingness to pay per fatality to estimate the benefit, the CEA could be transformed into a CBA and yield the same results. Until now, there was no allusion that the willingness to pay per unit to be used in order to estimate the benefits was not constant.

The dominant yardstick in all FSA studies that have been submitted to the IMO so far is the so-called “USD 3m criterion” which is the Implied Cost of Averting a Fatality (ICAF), as described in MSC 78/19/2. According to this, in order to recommend an RCO for implementation this must give a CAF value -both NCAF and CGAF- of less than USD 3 million. If this is not the case, the RCO is rejected. It has to be noticed that the CAF value (Cost of Averting a Fatality) is based on statistical analysis of the LQI (Life Quality Index) for OECD countries, see MSC 72/16, Skjong(2002) or Kontovas (2005) and this work for updated CAF values.

Note that according to the FSA Guidelines (MSC 83/Inf.2, Appendix 7) some approaches that can be used to estimate the NCAF and CGAF thresholds are :

1. Observation of the Willingness-To-Pay to avert a fatality;
2. Observation of past decisions and the costs involved with them; and
3. Consideration of societal indicators such as the Life Quality Index (LQI).

It is agreed many approaches may be used for the estimation of ICAF besides the one based on the LQI. Willingness to pay or to accept can provide the Value of a Statistical life (VSL) that can be used as an ICAF threshold. This will be briefly discussed in Section 3.7.

The next sections will describe the Life Quality Index (LQI). LQI is one of the so-called societal indicators which are statistics that quantify some aspect of the quality of life in a society or group of individuals. More specifically, social indicators are *"social statistics which represent significant information about the quality of life, and can be accumulated into a time series"* (see Nathwani et al 1997). The Gross Domestic Product (GDP) per person and the life expectancy (LE) are well known examples of social indicators.

3.3 The Life Quality Index (LQI)

The Life Quality Index (LQI) is intended as a social indicator that reflects the expected length of "Good Life", in particular the enhancement of the quality of life by good health and wealth. The original LQI definition is given by Nathwani, Lind and Pandey (see Nathwani et al,1997). A way of expressing it is as follows:

$$LQI = g^w \cdot e^{1-w} \quad \text{Eq. 3-2}$$

The ICAF value is determined by assuming that an option is accepted as long as the change in LQI owing to the implementation of the option (=RCO) is positive and under the assumption that the remaining lifetime in any given time is half of the life expectancy at birth . Therefore ICAF is given by as follows:

$$ICAF = \frac{g \cdot e}{4} \cdot \frac{1-w}{w} \quad \text{Eq. 3-3}$$

where

g is the Gross Domestic Product per capita

e is life expectancy at birth

w is the proportion of life spent in economic activity.

The concept of Life Quality Index (as societal indicator) was initiated at the Institute for Risk Research of the University of Waterloo in Canada. Among the first relative publications are Lind et al. (1991) and Nathwani et al (1997). The reader is referred to Pandey and Nathwani (2004) for a detailed explanation of the original deviation of LQI

and to Skjong and Ronold (1998), Rackwitz(2002) and Ditlevesen(2004) for more information on how to derive ICAF from the Life Quality Index. The literature on LQI is immense but it is out of the scope of this work to review them since the main focus of this Section is to arrive at an updated ICAF value by using the same methodology as before.

3.4 Implied Cost of Averting a Fatality (ICAF)

The dominant yardstick in all FSA studies that have been submitted to the IMO so far is the so-called “USD 3m criterion” which is the Implied Cost of Averting a Fatality (ICAF), as described in MSC 78/19/2. According to this, in order to recommend an RCO for implementation this must give a CAF value -both NCAF and GCAF- of less than USD 3 million. If this is not the case, the RCO is rejected. It has to be noticed that the CAF value (Cost of Averting a Fatality) is based on statistical analysis of the LQI (Life Quality Index) for OECD countries (see MSC 72/16, Skjong(2002) or Kontovas (2005) for updated CAF values) that will be discussed in the next Sections.

Coming back to the cost-effectiveness thresholds the above criteria are presented in the FSA Guidelines.

Table 3-1: Cost Effectiveness Criteria

	NCAF [US \$]	GCAF [US \$]
criterion covering risk of fatality, injuries and ill health	3 million	3 million
criterion covering only risk of fatality ^{*)}	1.5 million	1.5 million
criterion covering only risk of injuries and ill health ^{*)**)}	1.5 million	1.5 million

The proposed values for NCAF and GCAF in Table 6 have been derived by considering societal indicators (refer to documents MSC 72/16, and Lind, 1996). These criteria are based on the Life Quality Index (LQI) that was proposed by Nathwani, Lind and Pandey (Nathwani et al,1997). Actually, the value of \$3 million is based on the Implied Cost of Averting a Fatality (ICAF) and has been calculated using OECD data.

Skjong (2002) calculates the ICAF values (averages between years 1984 and 1994) for OECD countries. It has been proposed that the criteria of Table x should be updated every year according to the average risk free rate of return or using (approx. 5%) or by use of the formula based on LQI. In Kontovas (2005) an updated value was calculated using the same assumptions that were used by Skjong and Ronold(1998) and Skjong (2002) and the latest statistical data. The results were that the average ICAF value for

all OECD countries for the period of 2000-2002 is \$ 3.272 m whereas for the period of 1995-2002 is \$ 3.069 m . It should also be noticed that in the study of Skjong and Ronold data were given for 25 OECD member-countries while today these countries are more than 30.

Section 3.5 discusses the parameters that affect the value of the cost effectiveness threshold. In Section 3.6 an updated value given the latest available data is presented. Furthermore, note that according to Section 2.3, if the price of \$3 millions is used as the constant willingness to pay per fatality to estimate the benefit, the CEA could be transformed into a CBA and yield the same results. Until now, there was no allusion that the willingness to pay per unit to be used in order to estimate the benefits was not constant. On some thoughts regarding this issue see Section 3.7.

3.5 Parameters affecting the value of the current (LQI-based) CAF effectiveness index

Although any numerical value could be criticized, the need of a numerical criterion is essential and until now, the problem in the FSA process are not the exact numerical criteria but the way that costs and benefits are estimated for subsequent evaluation against the criteria. Furthermore, one could criticize the use of LQI in order to derive an ICAF value. This Section focuses on estimating an updated ICAF value using the latest available data and investigating the most important parameter of the LQI which is the proportion of life spent in economic activity, w (see Section 3.5.3). Although the need for further analysis on the non linearities of the damage function is well established, the ICAF criterion is well established in the relevant literature and therefore will not be easily abandoned by the researchers.

3.5.1 Gross Domestic Product per capita (g)

Gross domestic product (GDP) is a standard measure of the value of the goods and services produced by a country during a period of a year and as such the GDP per capita is an indicator of economic living standards. Often, the conversion into a common currency is made using exchange rates, but these give a misleading comparison of the real volumes of goods and services in the GDP. Comparisons of real GDP between countries can best be made using purchasing power parities (PPPs) to convert each country's GDP into a common currency in order to equalize the purchasing power of the different currencies.

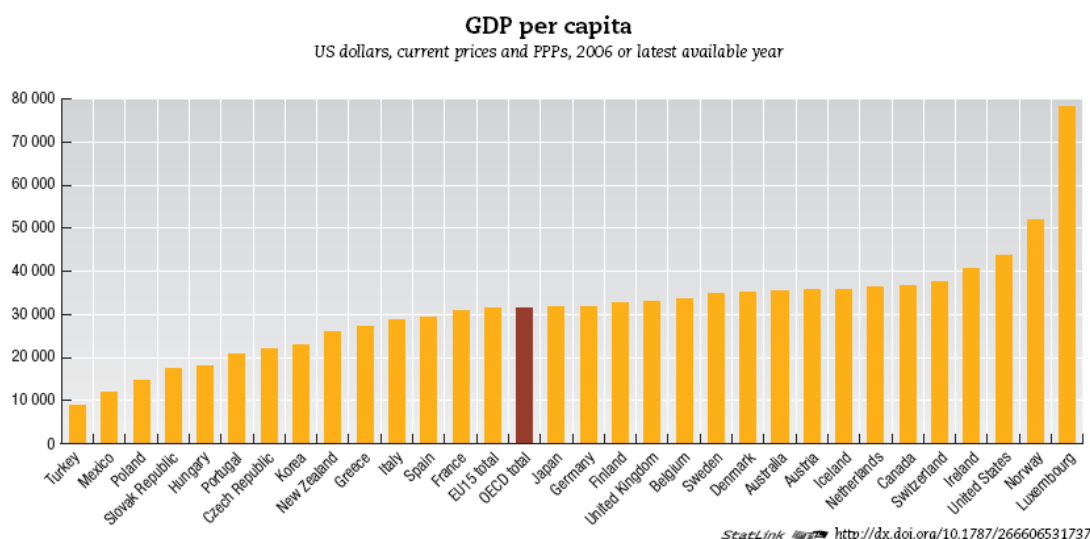


Fig. 3-1: GDP per capita (US dollars per capita per year) – Source: OECD Factbook(2008)

Some researchers suggest that only a percentage of GDP should be used in the calculations of ICAF. For instance, Rackwitz (2004) suggests that the GDP should be reduced to about 60% of the registered value stating that only a part of the GDP is available for decisions on how to consume the GDP. Furthermore, he notes that the GDP also creates the possibilities to “buy” additional life years through better medical care, more safety from natural hazards etc. While almost related studies use the full GDP we believe that Rackwitz is correct and that the matter worths further investigation. However in our calculations the full GDP will be used, the value of which for each OECD-member country will be derived from the latest OECD publication (see OECD Factbook(2008)) using the PPP method.

3.5.2 Life expectancy at birth (e)

Life expectancy, based on mortality statistics of nations, *“is a summary measure of the total mortality experience of a population which is little affected by any pattern of the age and sex pyramid, by the pattern of the birth rate or the history of migration”* (WHO 1977).

Life expectancy is the mean duration of life of a person of specified age and sex and can be used as a safety index. The LQI formula uses the life expectancy at birth that is the mean duration of life at the time of birth. OECD Factbook(2009) provides the life expectancy for both genders -which variates significantly in some countries- but the average life expectancy will be used in this study.

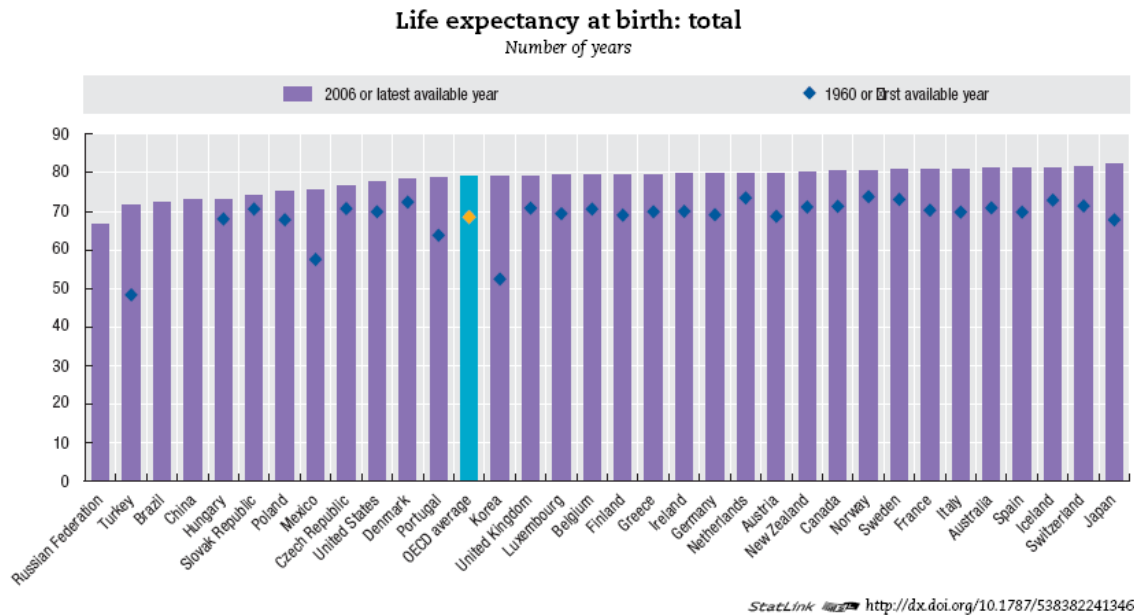


Fig. 3-2: Life expectancy at birth per country (years) – Source: OECD Factbook(2009)

A lifesaving action that prevents deaths among 5-year-olds has an impact higher, perhaps by an order of magnitude, than if it were applied to the 75 age group (Nathani et al, 2009). Nathani et al. (2009) also describe various discounting models that model the populations survival rate. According to Rackwitz (2002) in case of failure of a technical object the number of lost life years e_r in an event is between $0.67e$ (for young groups with a triangular age distribution) and approximately $0.5e$ (for aging groups) on average. The ICAF formula (see eq. 3-3) was derived under the assumption that the remaining life of an individual at any given point of time equals to half of the life expectancy e at birth (see Skjong and Ronold(2002) and Kontovas(2005)).

3.5.3 Proportion of life spent in economic activity (w)

This is the most controversial parameter in the estimation of the ICAF value. W is the proportion of life time that one spends in economic activity and as such depends on the country and on occupation. For example, in developed countries a 42 hours worktime per week is considered an average value, but the maximum hours of work in any seven-day period for seafarers can be up to 72 (International Labor Organization, Convention C180), see Kontovas(2005). According to Rackwitz(2002,2003) w has a value around 0.16 for developed countries but can go up to more than 0.20 for developing countries. Nathwani et al. (2002,2007) proposed an average value of w can be approximately taken as $1/8$ per year over the life span of an individual. According to Ditlevsen(2002), “a value of c of order of size 0.15, as currently preferred in the literature, puts too small weight on the money side” and proposes a value of 0.3.

Kontovas (2005) illustrates different values of ICAF for w varying from 0.1 to 0.3. We will use the same methodology in order to perform a sensitivity analysis of the calculation of ICAF. The work-time fraction is obtained as the total number of work hours divided by the national population. The total number of hours worked per year is the product of the number of employed people and the annual number of work hours per worker.

Most studies estimate w by using some assumptions of the working profiles. Kontovas (2005) assumes an expected life at birth to be 80 years; the first 18 years are assessed to be work free (last years of them occupied with education), the next 50 years of life are the working years (8 hours per day in 5 days per week in 45 weeks per year) and the last 12 are years of retirement. This gives a value of $w=0.129$. A more sophisticated profile is used by Nathwani et al(1997). By assuming that in North America, the “average person” works about 50 years out of 80 years of life, 48 weeks per year out of 52, and about 42 hours per week (including time spent travelling to and from work) out of 168 the arrive at a value of $w=(50/80)(48/52)(42/168)=0.144$. By further assuming that a considerable proportion of the time spent in economic production goes towards health care, the conclude that $w=(0.141)(1/0.101)=0.125=1/8$. This calculation takes into consideration that the total health care expenditure in Canada in 1995 is about 10.1% of gross domestic product.

Another way to estimate w is by using statistical data. In our analysis, w will be calculated using the OECD Factbook(2009) which provides the average hours worked as the total number of hours worked over the year divided by the average number of people in employment (see Fig.3-3). w will be calculated by dividing the average worked hours per year by 8,760 –that is the total number of hours per year by assuming 365 days per year and 24 hours per day, which is in line with Ditlevsen (2003).

Note that there is always some degree of uncertainty in statistical data, for example employment is measured through household labor force surveys. Furthermore, national statistics are not always comparable and based on a range of different sources of varying reliability.

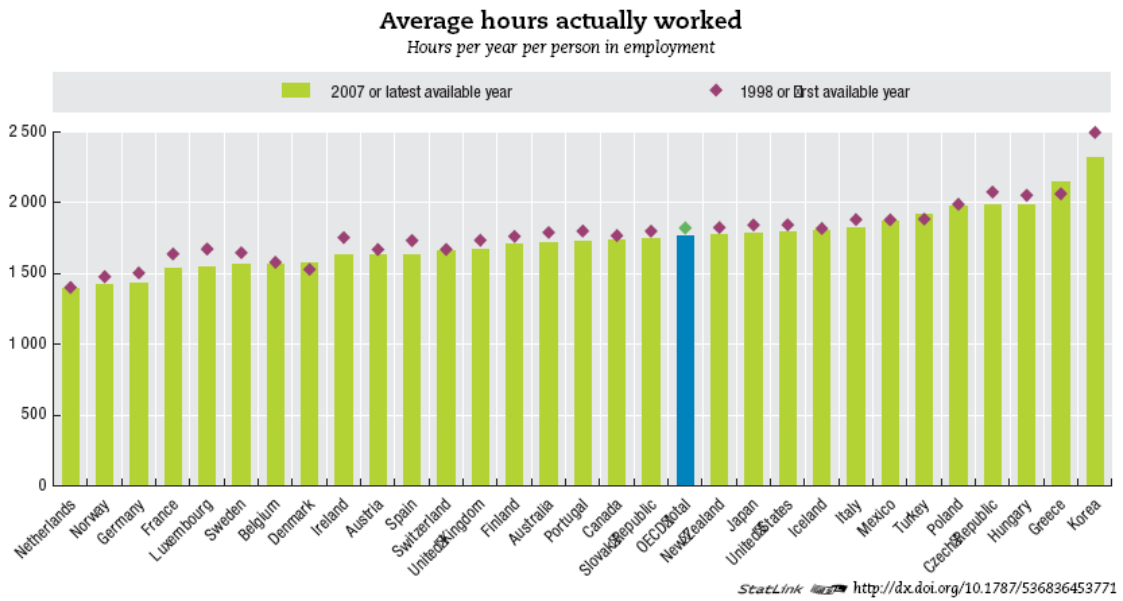


Fig. 3-3: Average hours worked per person in employment – Source: OECD Factbook(2009)

OECD Factbook(2009) includes values for year from 1970 through 2007. **The average value for all OECD countries for the last 10 years (which is also very close to the median) is equal to 0,201.** Based on the latest available data, for 2007, value varies from 0.159 for Netherlands to 0,264 for Korea.

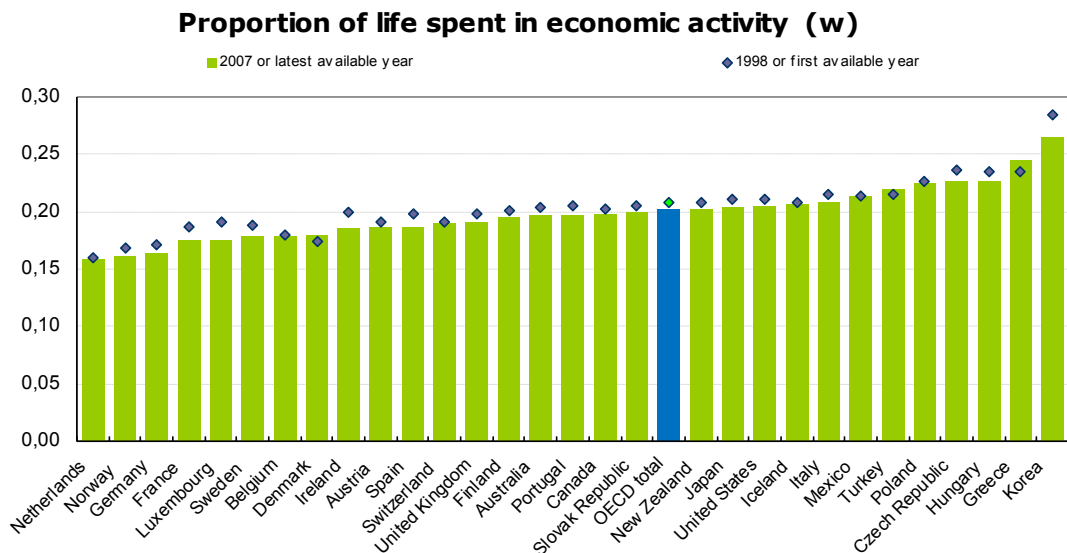


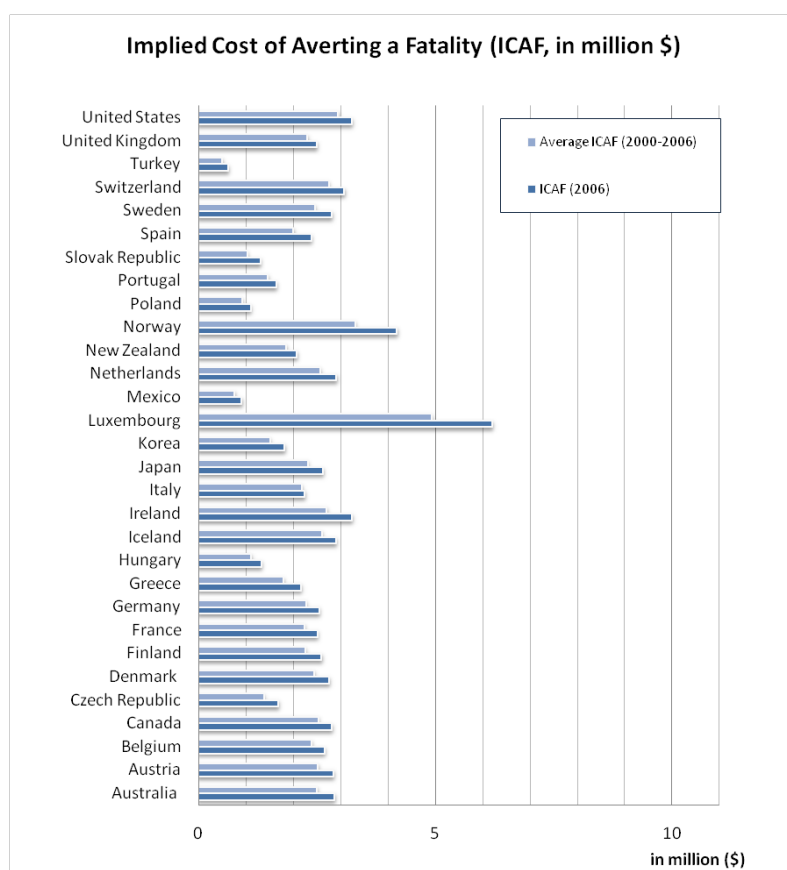
Fig. 3-4: Proportion of life spend in economic activity – Data from: OECD Factbook(2009)

3.6 Updated value for Cost of Averting a Fatality (ICAF)

By using Eq. 7 and the data for g, e and w as discussed in Sections 4.3.1, 4.3.2 and 4.3.3 we can estimate the ICAF value for all OECD-member countries. The ICAF values for all OECD countries (using a proportion of life spend in economic activity, w , equal to $1/8$) are illustrated in the following figure.

By setting factor 'w' equal to 0.125 the average ICAF value for 2006 is \$ 4,356,518.96 and for years 2006-2007 the average ICAF is \$3,767,031.24. Furthermore, by using the average value for all OECD countries for the last 10 years -that is $w = 0,201$ (as calculated in Section 3.5)- average ICAF value for years 2006-2007 the average ICAF becomes \$ 2,152,590.

We can also calculate the average ICAF value by using the time factor w as it is estimated for each country by using the **average hours worked per person in employment**. In this case, **for years 2000-2006 the average ICAF goes down to \$ 2,296,930, which is well below the \$3million yardstick being currently in use.**



Notes:1. Latest available data are for year 2006 (see OECD Factbook, 2009)
 2. Proportion of life spent in economic activity is set to $w=0.125$ as in relative studies

Fig 3-5:Updated ICAF Values – Data from: OECD Factbook(2009)

3.7 Non linearities in estimating benefits from reductions in the risk to humans

To begin with, it is obvious that both mortality and morbidity can be assessed. Numerous examples of how cost effectiveness analysis can be applied in the case of potential human loss have been presented, see for example the FSA studies submitted to the IMO. Note that the only technique used within FSA is CEA; probably as a result of the opposition of experts to monetize risk to human life. In addition, within FSA outcome is measured by the number of fatalities averted although in other industries the most widely used outcome measure is the quality-adjusted life-years (QALY).

On the other hand, within CBA of regulatory policies, the outcome measure is in most cases the so-called “value of a statistical life” (VSL). Without getting into more detail into this rather difficult subject, one of the ways to arrive at such a price is through Willingness to Pay (WTP) by answering the question “How much would individuals sacrifice to achieve a small reduction in the probability of death during a given period of time?” (Stavins, 2008). Similarly, the Willingness to Pay (WTP) is the compensation that individuals would accept for a small increase in that probability. Furthermore according to Stavins (2008), VSL may be viewed as the “marginal valuation for a small change in risk that is extrapolated to a risk change of 1”.

For example, if an individual is willing to pay \$3 for a risk reduction from 4 in a million to 3 in a million then the value of statistical life (VSL) is:

$$\text{VSL} = \frac{\$3}{4 \cdot 10^{-6} - 3 \cdot 10^{-6}} = \frac{\$3}{1 \cdot 10^{-6}} = \$3 \cdot 10^6$$

It is obvious that this does not mean that an individual is willing to pay \$3 millions to avoid a certain death. After all, asking such a question seems immoral. However this is also the sum of money that society would pay to eliminate a risk that is expected to kill one citizen.

In maritime risk assessment and more specifically within FSA, the cost effectiveness of risk reduction measures that assess safety of human life is assessed through the so-called “Cost of Averting a Fatality” (CAF). Any measure with an incremental cost effectiveness ratio less than the value of the so-called “Implied Cost of Averting a Fatality” (ICAF) should be recommended to be applied. In all recent FSA studies, cost effectiveness is limited to measuring fatality risk reduction using the \$3m criterion.

This criterion is to cover fatalities from accidents and implicitly, also, injuries and/or ill health from them. There are also two other criteria, one is to cover only risk of fatality and another to cover risk from injuries and ill health and both have a value of \$1.5m. The ICAF value of \$3 million can be interpreted "*as the economic benefits of averting a fatality*" (Kristiansen, 200x). This threshold is calculated based on the analysis of a social indicator, the so called Life Quality Index, see Skjong and Ronold (1998) and Kontovas (2005) for an updated analysis. We may argue that ICAF has essentially the same meaning as the value of statistical life.

Then, the question that may arise is why this cost effectiveness threshold or similarly this value does not depend on the risk reduction. For example, in all cases, a single per unit value is commonly acceptable within FSA studies when accessing risks to human life. We argue that the benefits of risk reductions measures are a non linear function of the risk reduction which in this case we measure by using the notion of potential loss of life. Note that in the case of human safety, risk reductions are, however, small.

In our opinion, these small figures contribute to this perception. The non linearity would be obvious if the amount of loss of lives averted were greater than 1, or if another unit was used. For example there is no doubt that mortality risk would not be assumed constant if it was not measured in a per life basis but let's say in a per day basis. Or to emphasize that this value applies only for small probabilities, like Howard (1980), we could think about the WTP or WTA as the amount of money per one-in-one-million probability of death or "micromort". Howard (1984, 1989) advocated the concept of a micromort ('micro' for millionth and 'mort' for mortality) to describe a one-in-a-million chance of death. Howard (1984) has also suggested other nomenclature, such as a "microdisability" for risks that lead to disability rather than death, we could even use "micromorb" for morbidity risk reductions (Cameron, 2010). If these tiny risk units were used it is obvious that WTP and WTA could not be constant even for small risk reductions.

To elaborate a little bit more on this we will present the case of morbidity. In health economics and in the FSA guidelines, the so called QALY as a measure of morbidity is presented. The Quality Adjusted Life Year (QALY) takes into account the quantity and quality of life that is saved by using a risk reduction measure. According to the FSA Guidelines (IMO,2007) "*the basic idea of a QALY is straightforward. It takes one year of perfect health-life expectancy to be worth 1, but regards one year of less than perfect life expectancy as less than 1.*" Within the scope of FSA, it is assumed that one prevented fatality implies 35 Quality Adjusted Life Years gained, see MSC 72/16.

Actually, it is widely accepted that the WTP per expected QALY is not constant across health risks and individuals. Hammitt of Harvard University has written on this issue over and over. *“Economic theory suggests that WTP should increase almost proportionately to the size of the risk change”* (Robinson and Hammitt, 2010). Hammitt (2005) states that *“you could think of willingness to accept compensation for a small increase in risk. For small changes, the slope is the same. It doesn’t matter if we think of willingness to pay or willingness to accept”* and Eeckhoudt and Hammitt (2001) that examine the effects of background mortality and financial risks on an individual’s willingness to pay to reduce his mortality risk (the value of statistical life or VSL) note that *“The effects of large mortality or financial risks on VSL can be substantial, but the effects of small background risks are negligible”*. Empirical studies have found that individuals state a constant WTP for small risk reductions and one possible explanation may be *“that individuals are indifferent between risk changes of these magnitudes”*. (Robinson and Hammitt, 2010).

Again, as discussed before in most risk analyses that assess the risk to human life there was never an allusion that a non linear “cost” function should be used. One reason is that risk changes of such small magnitudes are not well understood by researchers. However, there is no evidence that an individual may state the same WTP for risk reductions of lets say 1 QALY and 2 QALY. Similarly when accessing the risk of oil pollution the value of risk reduction in terms of tonnes averted are way larger and thus the use of non linear damage functions is viewed as the norm.

Another, main reason has to do with the fact that the \$3million figure is directly linked to cost effectiveness analysis where the use of a constant figure is indirectly implied. Johannesson (1995) writes that *“the difference between cost-benefit analysis and cost-effectiveness analysis is that in cost-effectiveness analysis the willingness to pay per QALY gained is assumed to be the same for all individuals under all circumstances and for all sizes of the change in QALYs.”* Recall from Section 2.3 that the only way to have a link between CEA and CBA is to use the same dollar per risk unit value of the outcome in the same sense that if the WTP per expected QALY was constant across health risks and individuals, then a cost benefit analysis that uses this constant WTP to measure benefits and a cost-effectiveness analysis that uses QALYs to measure health effects would produce equivalent results (Johannesson, 1995; Garber and Phelps, 1997).

4

THE ECONOMIC EFFECTS OF OIL SPILLS

While it is generally accepted that the overall level of maritime safety has improved in recent years, further improvements are still desirable. The same is also true as regards the level of protection of the marine environment. In the last decade, the number of oil spills and the total quantity of oil spilled in the seas have declined as it will be shown below. In spite of this positive development, one would like to know how the cost components associated with oil spills behave, and this Section attempts to shed some light into this question. A first step is to analyze the sources of oil spills.

4.1 Introduction to Oil Spills and its Sources

Since 1974 the International Tanker Owners Pollution Federation (ITOPF) has maintained a database of more than 10,000 oil spills from tankers, combined carriers and barges. The ITOPF database includes all reported accidental spillages (except those resulting from acts of war) which are divided into three categories by size (less than 7 tonnes, between 7 and 700 tonnes and greater than 700 tonnes). It should be noted that the figures for the amount of oil spilt in the following analysis present the total oil lost to the environment (including the oil one that is burnt and the one that remains in a sunken vessel). According to ITOPF (Huijer (2005) and ITOPF(2009)) the majority of reported spills are from small operational spillages of less than 7 tonnes for which complete reporting is difficult and, thus, *“little statistical reliance is placed on the data”*.

Table 4-1 presents the number of incidences of spills categorized by cause for the period 1975-2008. Consequently, Figure 4-1 visualizes the data from this table by showing the percentages of each cause for the spill size categories and for spills over 7 tonnes.

Table 4-1: Incidence of Spills by Cause, 1974-2008- Source: ITOPF (2009)

CAUSE	<7	7-700	>700	>7	TOTAL
OPERATIONS					
Loading/Discharging	2825	334	30	364	3189
Bunkering	549	26	0	26	575
Other operations	1178	56	1	57	1235
ACCIDENTS					
Collision	175	303	99	402	577
Grounding	238	226	119	345	583
Hull Failure	576	90	43	133	709
Fire/Explosion	88	16	30	46	134
Other	2188	152	26	178	2366
TOTAL	7817	1203	348	1551	9368

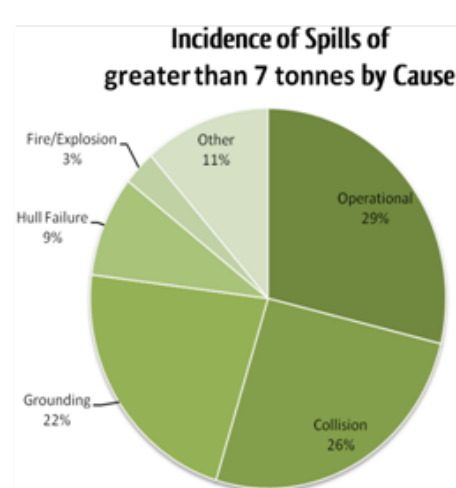
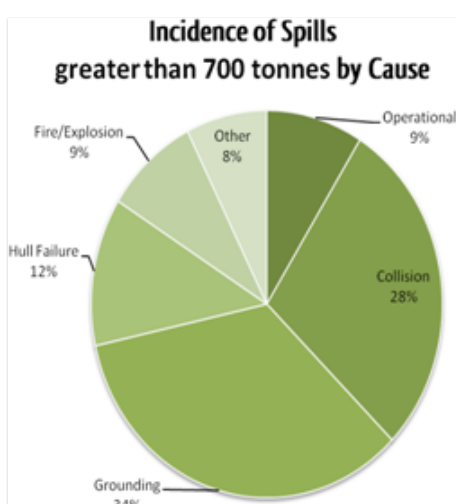
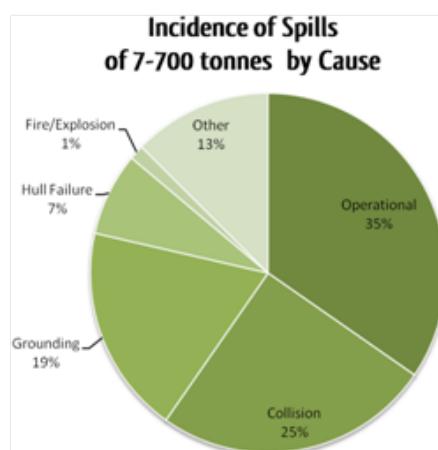
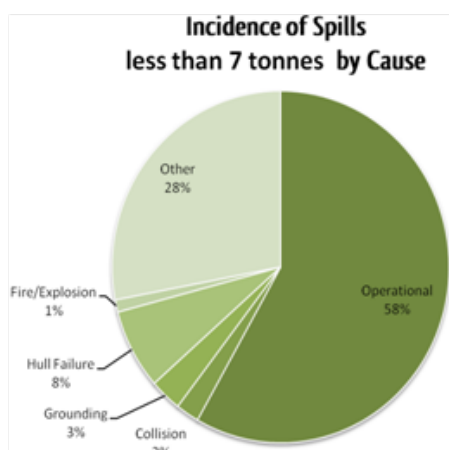


Fig.4-1: Percentage of Incidence of Spills by Cause, 1974-2008 (based on data from ITOPF(2009))

It is apparent from the above figure that most spills from tankers result from routine operations, which normally occur in ports and oil terminals. The majority of these spills are small (less than 7 tonnes). Of the accidental causes, most incidents are from collisions and groundings. These results are in line with Huijer (2005) and Burgherr (2007). In addition, various casualty analyses have been performed for oil carriers, see for example Eliopoulou and Papanikolaou (2007) and Papanikolaou et al. (2007). In addition, the risk of oil pollution from ship accidents has been also addressed among others in Giziakis and Bardi-Gkiziakis (2002), Ventikos and Psaraftis (2003), Papanikolaou et al. (2010).

However, we should stress that the categories considered (Collision, Explosion/Fire, Grounding, Hull/Structural Failure and Other) are the same with those used by Lloyds' LMIU, IMO, ITOPI and other casualty databases. Many such databases are more useful for aggregate statistical analysis of casualty data and less useful to draw conclusions as to the real cause of an accident and the sequence of events related to it. The latter may actually be a complex task to ascertain, as it may be the object of an accident investigation that may take years to complete, not to mention that it may be the outcome of a litigation process that can be equally as long. Another drawback of databases such as the above is that root cause information is usually missing, as such information can only be retrieved after considerable analysis of the accidents themselves. Working with casualty databases that have incomplete or even wrong cause information may skew the ensuing analysis, particularly regarding measures to reduce risk.

The lack of sufficient and relevant casualty data and the lack of data on the root causes of accidents in the existing databases has been repeatedly stressed, see Kontovas and Psaraftis (2009). For example, databases categorize casualties as collision, grounding, fire, explosion, foundering etc. But these are consequences, not causes. Something always happened before a grounding or a collision occurred, and focusing on the consequence and not the root cause may skew the analysis that follows, and (at minimum) shift regulatory focus to measures that mitigate the consequence rather than prevent the cause.

Lack of real cause information in casualty databases remains a serious hindrance to researchers and risk analysts. Because of unclear or misleading cause information, the recommended Risk Control Options (RCOs) tend to be geared toward mitigating the consequences of accidents instead of preventing the accident in the first place (i.e. RCOs may be proposed so that the vessel better withstands a collision). However, if sufficient data was available for the root cause of the collision, better and possibly

much more cost effective RCOs could be proposed (hypothetically duplication of a small mechanical part whose failure may lead to loss of power / steering). Most importantly, the accident could be prevented.

In this context, we draw attention to recent IMO doc. MSC 86/19/1 by Germany, and to doc. MSC 86/17/1 by Greece, that make this important point (among others), a point which has (to our knowledge at least), received far less attention than what it deserves. For a discussion of issues pertaining to uses of casualty databases, see also Devanney (2008).

Getting back to the analysis, as we discussed before, the majority of accidental spills on ITOPF’s dataset fall into the category of spills less than 7 tonnes. Furthermore, the number of spills per year for accidental spills of less than 7 tonnes is not presented by ITOPF. Figure 4-2 shows the number of spills per year and, as one may notice, the dominant size category is that of 7-700 tonnes. It is also apparent that there has been a decrease in the total number of spills as we will prove later on.

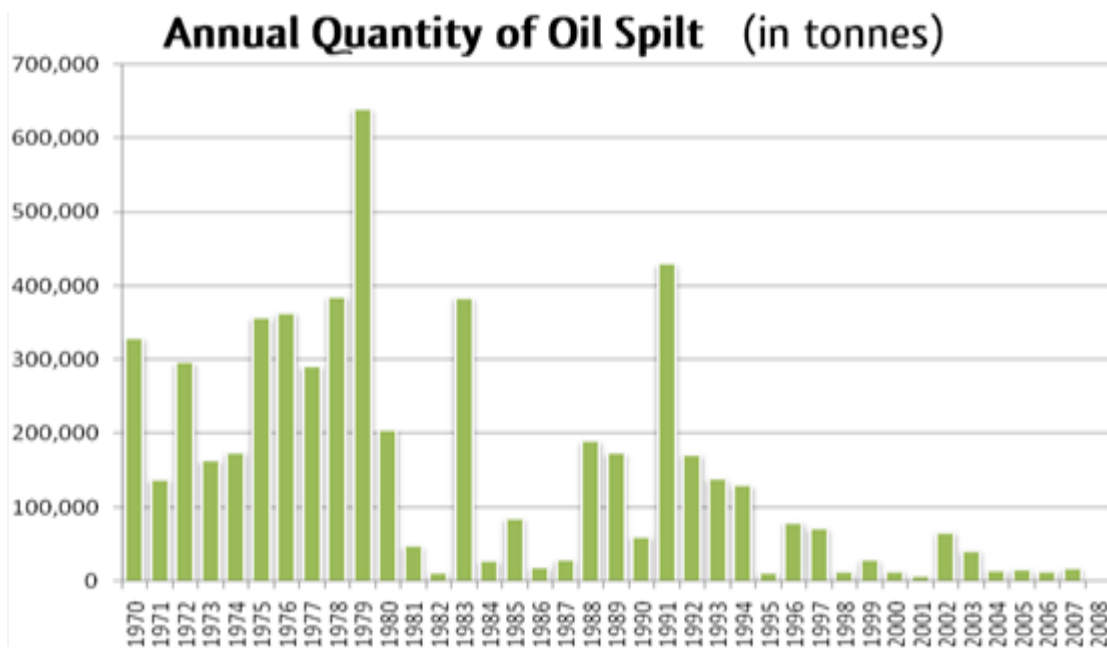


Fig.4-2: Annual Quantity of Oil spilt, 1974-2008 - based on data from ITOPF(2009)

The same downward tendency is apparent in the total annual quantity of oil spilt during the last decade (see Fig. 4-2). After the accidents of tankers Heaven (144,000 t) and ABT Summer (260,000t) in 1991, no accident above 100,000t has happened and, thus, the total amount of oil spilt decreased continuously. Figure 4-3 presents the average spill size per year and as one may notice after 1991 all average spill sizes are below 4,000 tonnes.

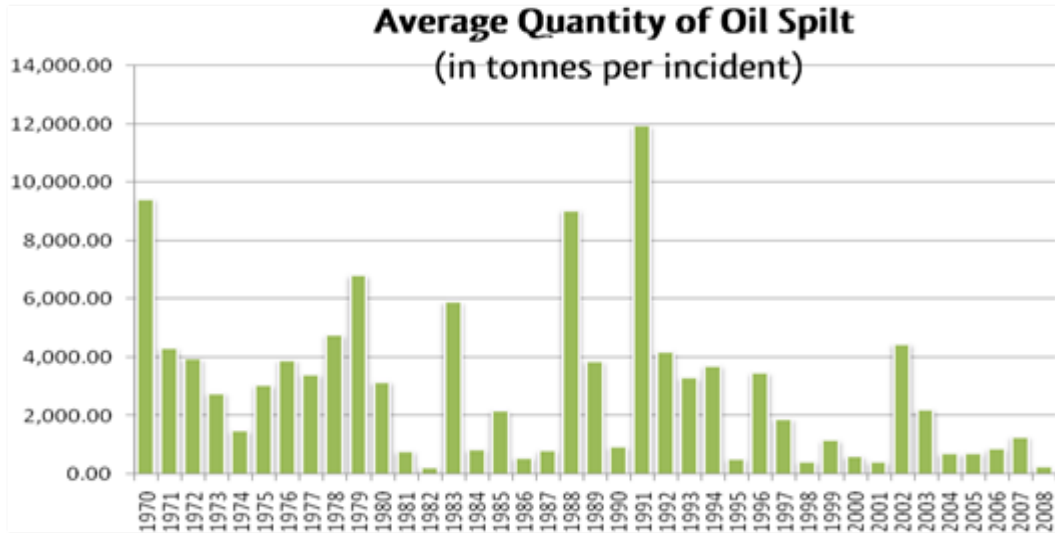


Fig. 4-3: Average Quantity of Oil spilt per Incident, 1974-2008 - based on data from ITOPF(2009)

According to ITOPF(2009) *“the incidence of large spills is relatively low and detailed statistical analysis is rarely possible,”* thus, emphasis is only given on identifying trends (see Huijer (2005) and Burgherr (2007)). Burgherr (2007) provides an in-depth analysis of accidental oil spills from tankers if oil spills above 700t for the period of 1970-2004. Besides the trends identified in that paper, a global analysis addressing spatial autocorrelation of oil volumes across all locations using Marsden Squares (10° latitude by 10° longitude) was performed to detect if data are clustered or distributed randomly in space. Furthermore, the distribution of tanker spill volumes, and more precisely, the contribution of large spills to total volume was investigated using Lorenz curves. Supporting the previous comment by ITOPF the only statistical analysis that makes real sense is the identification of a possible trends.

Thus, statistical analyses were performed to determine a possible trend included the calculation of the Autocorrelation Function (ACF) and Kendall’s τ for the three time series (see Fig. 4-4). The analysis regards the last 30 years (period 1978-2008).

As it can be seen in Appendix A, the results of the Mann-Kendall test (under the alternative hypothesis) at a 95% confidence level display negative Z values. In plain English this means that there is enough evidence for the existence of a downward trend in all three categories for the period 1978-2008 (see Table 4-3). Furthermore, all p-values are close to zero, which means that the trend is not caused by random sampling.

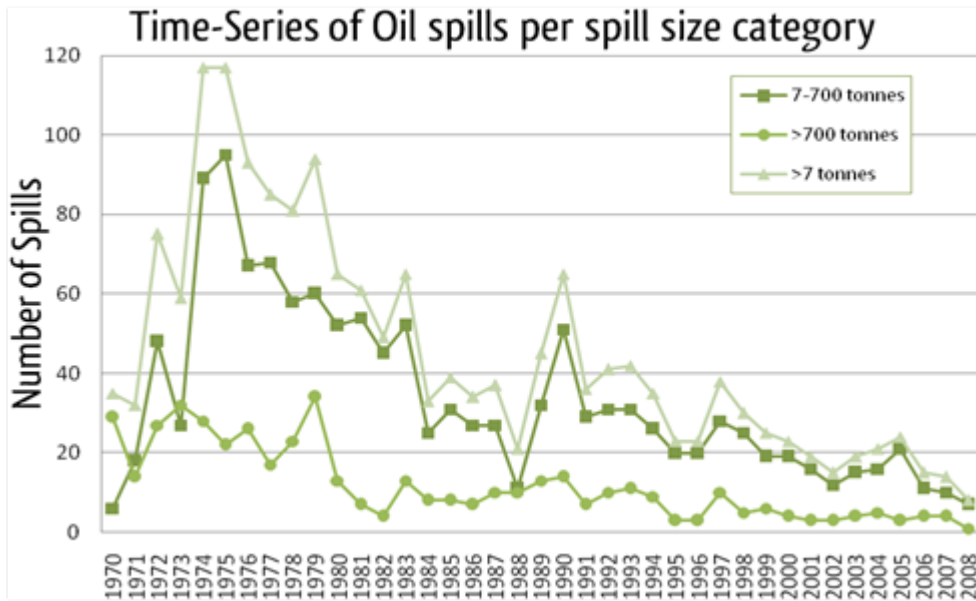


Fig. 4-4: Number of Oil Spills per Year, 1970-2008 - based on data from ITOPF(2009)

As discussed above, the number of oil spills per year as well as the total amount of oil spilt are constantly decreasing in the last three decades. However, the reduction of oil pollution is one of the stated goals of new regulations, including the implementation of double hulls for tanker vessels.

4.2 A Cost Effectiveness Criterion for oil spills

We now come back to the issue of how such environmental criteria can be used within FSA. A major topic in Annex 3 of doc. MEPC 55/18 and also in a report by EU research project SAFEDOR (Skjong et al, 2005; Vanem et al, 2007a) was the definition and analysis of risk evaluation criteria for accidental releases to the environment, and specifically for releases of oil. To that effect, the criterion of CATS (for “Cost to Avert one Tonne of Spilled oil”) was defined as an environmental criterion equivalent to CAF, “Cost to Avert a Fatality”. According to the CATS criterion, a specific Risk Control Option (RCO) for reducing environmental risk should be recommended for adoption if the value of CATS associated with it is below a specified threshold, otherwise that particular RCO should not be recommended.

$$CATS = \frac{\Delta C}{\Delta R} \quad \text{Eq. 4-1}$$

where

ΔC is the cost per ship of the RCO under consideration.

ΔR is the risk reduction per ship, in terms of the number of tonnes of oil averted, implied by the RCO.

The only case until now that this index has been used is the FSA on crude oil carriers prepared by SAFEDOR. Note, that this index is nothing more than the usual ICER index as presented in Section 2.

In the SAFEDOR report (Skjong et al., 2005), a threshold of USD60,000 per tonne of spilled oil was postulated for CATS, based on a series of modelling and other assumptions that will be presented shortly. It should be stressed out that this high value and the reactions by some States opened the issue of non linear functions which stimulated the author's work on this issue, see next Chapters.

4.3 Valuing oil Spills – Oil Spill Cost components

Even though the discussion at the IMO on environmental risk evaluation criteria for FSA has started only recently, the subject itself is not new, and substantial work has been performed over at least the last 30 years, mostly in the context of analyzing the economic impact of oil spills and contemplating measures to mitigate their damages. We note that an important part of this work concerns oil spill damage assessment. Among many other researchers, White and Nichols (1983) reported on the various components of the oil spill costs and on the significant difficulties in estimating these costs. Grigalunas et al (1986) reported on the socioeconomic costs of the AMOCO CADIZ oil spill (1978, France). In the context of the 'MIT oil spill model', Psaraftis and his colleagues at MIT used a 'damage assessment model' to estimate the damages of an oil spill in the context of optimizing oil spill response alternatives. They used damage cost estimates for various strategic spill response scenarios in the US New England region that ranged from about 29,000 USD/tonne (1983 dollars) for very small spills that typically occur close to shore to less than 300 USD/tonne for very large offshore spills (Psaraftis et al, 1986). More recently, the work of Etkin (1999, 2000, 2001, 2004), White and Molloy (2003), Shahriari and Frost (2008), and others provide significant material as regards both the methodology to compute oil spill costs and actual numbers to document these costs.

According to Liu and Wirtz (2006), five different categories of costs can generally be identified that can be divide into three groups: cleanup (removal, research and other costs), socioeconomic losses and environmental costs. By adding up these three cost categories one can obtain the total cost of an oil spill. Beyond any doubt, the cost of an oil spill is very difficult to estimate.

The total cost of an oil spill can be derived by using at least four different methods. These are the following:

1. Adding up all relevant cost components (cleanup, socioeconomic and environmental).
2. Estimating the clean-up costs through modeling and then assuming a comparison ratio for environmental and socioeconomic costs. Vanem et al (2007a) assumed a ratio of 1.5 and according to Jean-Hansen (2003) environmental costs, including socioeconomic costs, are almost 2 times the cleanup costs in Norwegian waters. Analysis of the IOPCF dataset suggests a ratio of less than 1, see Kontovas et al. (2010).
3. Using a model that estimates the total cost such as the Etkin BOSCEM approach
4. Assuming that the total cost of an oil spill can be approximated by the compensation eventually paid to claimants. Compensation information is reported by the International Oil Pollution Compensation Funds (IOPCF) which publishes annual reports. These have been used by Grey (1999) and, recently, by Yamada and Kaneko (2007). The latter was submitted to IMO and is to be presented at MEPC 58 in October 2008. This approach will be discussed in Section 7 of this paper.

4.3.1 Removal, Research and Other Costs

The International Tanker Owners Pollution Federation (ITOPF) has presented a description of the fate of an oil spill. When spilled at sea, oil normally breaks up and is dissipated or scattered into the marine environment as a result of a number of processes that change the compounds of oil. Thus, there is a general agreement (Etkin,1999; Grey, 1999; White and Molloy, 2003) that the main factors influencing the cost of oil spills are:

1. Type of oil

Moller et al (1987) found that cleanup costs for light oils and refined products tend to be below the average cost. Light products, in most of the cases, are more toxic than heavier oils, however, they disperse more readily. For example, according to an analysis of cleanup costs of US and non-US spills by oil type (Etkin,1999) the average cleanup cost for light crude oil is 4,265.94 USD per tonne while when involving Marine Heavy Fuel Oil (MFO) the cleanup cost is 23,893.28 USD per tonne.

2. Location

A spill occurring far from the coast tends to cause minor damages as oil will be dispersed before reaching the shore. The ATLANTIC EMPRESS accident off the coast of Tobago in 1979 was the reason of a 280,000 tonnes spill but caused little damage because of its location and also due to favorable wind and weather conditions.

3. Weather and sea conditions

Obviously, favorable wind can prevent the oil from reaching the shore which could lead to higher costs. Furthermore, good weather would result in a more rapid clean-up process. Added to this are the limitations on oil collection systems imposed by bad conditions such as wind, waves and currents.

4. Amount spilled and rate of spillage

There is definitely a relation between the costs of a spill and the amount of spilled oil. In general, larger spills imply higher costs but the relation is not linear as shown by Etkin (1999) who came to the conclusion that the clean-up costs on a per tonne basis decreased significantly with increasing amounts of oil spill. White and Molloy (2003) have discerned a similar trend in their analysis using ITOPF's data and insist that simple comparisons between the costs of individual spills based only on a per volume unit can be highly misleading.

Furthermore, the rate of spillage is also an important factor because, for example, the clean-up operation required in response to a single spill may be considerable but will be completed in a matter of days or weeks. However, the same quantity if lost over several months require repeated cleaning and will have long-term effects. This was the case of BETELGEUSE, a tanker that sank at a terminal in Ireland and because of the ongoing release from the various parts of the wreck the clean-up respond lasted for some 21 months although the total amount of oil spilled wan no more than 1,500 tonnes.

5. Clean-up Response

Quite understandably, as an immediate response to an oil spill, all the effort is devoted to deal with the spilled oil in an attempt to prevent the damage and the public outcry - which is mostly associated with pollution of shorelines. In most of the cases, well-organized operations and rapidity of response are fundamental to limit the clean-up costs. The management of response operations is being extensively discussed in White and Molloy (2003).

One of the early studies on oil spill cleanup costs was performed by Cohen (1986). For example, based on data owned by the USCG (regarding 95 accidents between 1973 and 1981) he proposed the use of a non linear function that uses the volume of oil spilled and some other parameters that depend on the location of the spill. Later, Etkin (1999) devised a method for estimating clean-up costs (on a per tonne of oil recovered basis) based on location, shoreline oiling, type of oil spilled, cleanup strategy and amount spilled. She further refined the model by adding two more variables: the specific type of location (allowing for three type of spills: offshore, coastal and port spills) and the country location. This new model by Etkin (2002) was based on a number of spills that happened worldwide while her previous models were based on US spills only. Her analysis (Etkin, 2001) showed that average costs could vary by at least one order of magnitude. Thus, the average clean-up cost (in 1999 USD per tonne) for an oil spill in Lithuania is 78.12, in Malaysia 76,589.29 and 25,614.63 in the United States. Finally, Shahriari and Frost (2008) have, very recently, developed a mathematical method to estimate cleanup costs based on regression analysis of 80 incidents during the period 1967-2002. The model parameters are spill quantity, oil density, distance to shore, cloudiness (used as a measure of how much sunlight

reaches the oil which is the main factor that affects evaporation) and level of preparedness based on ITOPF estimations on how well different world regions cope with oil spills.

4.3.2 Socio-economic Losses

According to Liu and Wirtz (2006) socio-economic losses consist of property damage and income losses. The property damage can be estimated by adding up all costs of repairing or cleaning facilities including vessels. On the other hand, the income losses take into consideration damages from various sectors such as fishery and tourism. The total economic losses are the sum of foregone incomes during the recovery period. This part of the total cost is very straightforward to estimate and needs no more explanation.

As regards this category, one thing is clear: This is not an easy subject. It is clear that the value of lost oil should count as part of the damage cost of a spill (and this is the easiest part to compute). Also, income lost by fishermen in the vicinity of a spill should be counted as part of the socioeconomic cost of that spill. The same is true for income lost by hotels, restaurants, and other tourist shops whose turnover is reduced as a result of a spill in their area. But what if tourists spend money in a restaurant to which they came to dine in an excursion to take a look at the spill? Should this count as a plus? Even attempts to calculate lost income as a result of people having a lower IQ because they systematically ate shellfish contaminated by oil have been recorded (see for instance INTERTANKO's comment in doc. MEPC 58/17). All this points out that estimating socioeconomic spill costs is generally very difficult and is never likely to be an exact science.

4.3.3 Environmental Costs

This part of the total cost of an oil spill is the most difficult to evaluate since most of environmental goods or services are non-market. Economists have developed a range of approaches to estimate the economic value of non-market impacts. In order to measure environmental damages economists either indirectly link environmental resources to some market goods or even construct a hypothetical market in which people are asked to pay for these resources. It is out of the scope of this paper to analyze these methods, however, one of them has been used in order to estimate the damages from the 'Exxon Valdez' oil spill and some other spills and will be presented in this section.

There was a rapidly growing interest in passive-use values in the US which was heightening at the time of the study by the passage of the Oil Pollution Act (OPA) and the regulations that National Oceanic and Atmospheric Administration (NOAA) enacted under it for natural resource damage assessments. The regulations state the *“the trustees should have the discretion to include passive use values as a component within the natural resource damage assessment determination of compensable values”*.

The Contingent Valuation (CV) method is a widely used non-market (or passive use) valuation method especially in the areas of environmental cost-benefit analysis (CBA) and environmental impact assessment (EIA). CV is a survey approach designed to create the missing market by determining what individuals or households are willing to pay (WTP) for specific changes in quantity or quality of environmental goods or, more rarely, by asking responders for their willingness to accept (WTA) in compensation for a specified degradation in the provision of these goods (Hanemann, 1999). The name for this form of valuation arose because the elicited values are contingent upon the particular scenario described to survey respondents.

An important benchmark in the history of the CV is that of the ‘Exxon Valdez’ oil spill. The oil spill due to the grounding of the oil tanker ‘Exxon Valdez’ in the Prince William Sound on March 24, 1989 was the largest oil spill from a tanker in US history, which affected more than 1,300 kilometers of coastline and caused the death of 23,000 birds. After the oil spill, the State of Alaska appointed an interdisciplinary group of researchers to design and implement a national CVM study to measure the loss of non-use values to US. This study was coordinated by Richard Carson and constitutes one of the major contingent valuation applications and represents an important methodological reference for all contingent valuation researchers' work. The loss of non-use values resulting from the ‘Exxon Valdez’ oil spill was estimated at 2,8 billion dollars (Carson, 1992). As a reaction to this study Exxon commissioned a group of researchers to verify whether non-use values could be accurately measured by means of CV. The main argument of critics of CVM is that this method is not capable of resulting in valid and reliable monetary measures of non-use values. Hausman’s well-know argument *“is some number better than no number”* fully expresses the skepticism toward this method. Therefore, according to Hausman, assessments of lost non-use values by means of the CVM method should not be used in court (Diamon and Hausman, 1994). In order to address the criticism, National Oceanic and Atmospheric Administration (NOAA, 1993) set a group of experts, with Nobel laureates Kenneth Arrow and Robert Solow as chairmen, in order to evaluate the reliability of the use of CVM in the natural resource damage assessments.

Despite the criticism, Contingent Valuation is the most popular and the most controversial of the methods that environmental economists use to value environmental goods and services and has been used to assess the impacts of many oil spills from tankers such as the 'Exxon Valdez' (Carson et al, 1992; 2003) , the 'Nestucca' and, very recently, the PRESTIGE (Loureiro, 2007). However, nowadays, the most commonly applied method especially by the National Oceanic and Atmospheric Administration (NOAA) in the United States is the so-called Habitat Equivalent Analysis (HEA). This method is specifically designed to determine the compensation the public is due to reconcile injuries to the ecosystem and the lost services that the ecosystem provides to the biotic component. According to the 1996 final rule of the Oil Pollution Act (OPA 90), "*when injured resources and/or services are primarily of indirect human use (e.g., species habitat or biological natural resources for which human uses are primarily off-site) the appropriate basis for evaluating and scaling the restoration is Habitat Equivalency Analysis (HEA)*" (King, 1997). The principal concept underlying HEA is that the public can be compensated for past losses of habitat resources through habitat replacement projects providing additional resources of the same type. The reader can find more information on this topic in NOAA (2000).

4.4 The cost per unit of amount of oil spilt – The CATS threshold

We now come back to the issue of how such environmental criteria can be used within FSA. As discussed above, in the SAFEDOR report (Skjong et al, 2005), a threshold of USD 60,000 per tonne of spilled oil was postulated for CATS, based on a series of modelling and other assumptions. In order to clarify how the authors arrived at this single value some more recent papers of the same authors will be addressed.

Vanem et al (2007a, 2007b) adjusted (in accordance with the changes in US Consumer Price Index) to 2006 dollars the regional average cleanup costs presented by Etkin (2000). These costs were weighted according to oil tanker traffic density distributions derived from the AMVER data for 2000-2001 (Endresen et al., 2004) and arrived at a world average cleanup cost of 16,000USD per tonne (see Table 4-2 below).

Table 4-2: Average Cleanup Costs [Vanem et al. 2007a]

Region	USD/tonne	Traffic share (%)
Middle East	1,300	7
South America	3,800	18
Africa	3,900	18
Oceania	6,900	2
Europe	13,100	11
North America	24,000	19
Asia	33,300	24
<i>Weighted average</i>	<i>15,900</i>	<i>100</i>

Finally, taking into account the work of Jean-Hansen (2003), McCay et al (2004) and Etkin (2004) they concluded that a ratio of 1.5 should be assumed for socioeconomic and environmental costs as compared to cleanup cost. Thus, the total oil spill cost is 2.5 times the cost of cleanup, which means 40,000 USD per tonne of oil spilled.

The implementation criterion that was proposed is the one presented by DMA and RDANH (2002) according to which the following fundamental approach is valid for measures implemented on ships:

$$\text{Cost of averting a spill} < F \times \text{Cost of an occurred spill}$$

It was suggested that risk reduction measures are to be implemented if the costs of averting a spill are less than the costs of an occurred spill multiplied by F, where F is an “assurance parameter” postulated to be between 1 and 3 ($1 < F < 3$). According to the above authors, this parameter, reflects the fact that spending resources on preventing oil spills is preferable to spending the same resources in the aftermath of a spill and recommended that a factor of 1.5 is a good one to be used in a global criterion. Note that in another paper of the same authors this value is said to be in line with the cost effectiveness of the OPA 90 regulations (Vanem et al., 2008).

To sum up, according to SAFEDOR the average global cleanup cost is 16,000USD per tonne, plus 24,000 USD per tonne to cover the socioeconomics and environmental costs, giving a total of 40,000 USD per tonne of oil spilled. Then, applying the 1.5 assurance factor they arrive at a CATS threshold value of 60,000 US dollars per tonne.

4.4.1 Critique of the \$60,000 CATS threshold

Kontovas and Psaraftis (2006) were probably the first to question the SAFEDOR approach, both on the use of any single dollar per tonne figure and on the 60,000 dollar threshold.

In fact, various spill cost data over the years suggested the following average cleanup costs worldwide (USD/tonne, 1999 dollars): 6.09 (Mozambique), 438.68 (Spain), 3,082.80 (UK), 25,614 (USA) and even the extreme value of 76,589 for the region of Malaysia (Etkin, 2000). The EXXON VALDEZ 37,000-tonne oil spill had a cleanup cost of 107,000 USD/tonne (2007 dollars), whereas the cleanup cost of the BRAER 85,000-tonne oil spill was as low as 6 USD/tonne. In addition, there is ample reference in the literature (see for instance Etkin (1999), among others, and even in Annex 3 of MEPC 55/18 itself and Vanem et al (2007a, 2007b)) that the cost of oil spills on a dollar per tonne basis depends on a variety of parameters and has a broad variance.

We also note that an implicit assumption in the weighting scheme of Table 4-2 is that regional oil traffic share (however that is defined) is an appropriate weight with which to multiply regional cleanup cost. The direct way to compute global average cleanup cost would be to divide global total cleanup cost by total tonnes spilled globally, that is, divide the sum of the products of the average cleanup cost in each region times the volume of oil spilled in that region by the sum of the tonnes of oil spilled regionally. Using oil traffic share in each region as the weight in Table 4-2 implicitly assumes that the total volume of oil spilled in a region is proportional to the total oil traffic through that region. However, this assumption may not be true, as certain regions may spill more than their traffic share, and others less. This is a product of different environmental conditions, different regulatory regimes, perhaps different technologies (ships, traffic control schemes, etc), or just the statistical behavior of oil spills, given that most of the oil is spilled in a handful of very large spills. For instance, it could be speculated that oil spill volume in North America is probably lower than that in Africa for the same level of oil traffic. In fact, the poor statistical correlation between total volume of oil spilled in a region and regional oil traffic has been documented long time ago, among others, in Devanney and Stewart (1974), who argued that finding an appropriate “exposure variable” for the distributions of the number and volume of spills is certainly a non-trivial subject. Computing global average cleanup cost by the direct way would change the weighted average of Table 4-2 (and in our opinion downwards).

But even if a single global cleanup average value could be commonly accepted, the ratio of 1.5 that is assumed to account for socioeconomics and environmental costs as compared to cleanup costs seems unsubstantiated. And, finally, if an ‘assurance parameter’ F (different from 1) is introduced, its appropriate value should only be ascertained after a quantitative assessment of society’s willingness to pay to avert oil pollution. By contrast, the value of F in the CATS approach was inferred ‘in reverse’, that is, chosen so as to certify that previous legislative action (in this case, OPA 90) to prevent pollution had been correct. Note that even Vanem et al (2007) state that “reservations should be made regarding the exact values that are suggested” (referring to the 40,000 USD per tonne figure) and that “when new and updated cost statistics become available the criteria should be modified accordingly”.

It is also noted that the CAF criterion, as currently applied in FSA, uses no F factor, or implicitly assumes $F = 1$. But one could make a similar (or an even stronger) argument with the one used for CATS, that one would be willing to pay a cost higher than the estimated economic value of human life to save a fatality. The question why one

should use the F factor for the environment whereas it is not used for human life is one that needs to be answered.

Leaving aside the issue that F, if used, should be determined by society or the maritime policy-makers and not by FSA analysts, it comes as no surprise that neither the postulated upper bound of F (3), or its lower bound (1) are necessarily valid. For instance, how can one be sure, beyond any reasonable doubt, that F is absolutely below 3? Society (or maritime policy-makers) may conceivably decide to spend 4 times as much upfront in the form of capital or other costs, so as to avert a given expected spill cost. In Psaraftis et al (1986), a weighting factor as high as 15 between damage costs and system costs was used to investigate strategic spill response alternatives, but no attempt to estimate what this weight might be was made.

That F should above 1 beyond any reasonable doubt is also debatable. Society may very well prefer to pay whenever oil spills occur, instead of paying an amount equal to the expected cost of these spills upfront. Also, and as those stakeholders who will bear the burden of cleanup and environmental costs are not the same who will pay for measures to prevent oil spills, the whole issue of the F factor is much more difficult than it appears in the first place.

Kontovas and Psaraftis (2006) was the core of a submission by Greece on this issue (doc. MEPC 56/18/1). The main thrust of Greece's position, pointing out the deficiencies of basing cost calculations on spill volume, was by and large supported by various arguments by the United States, the International Association of Independent Tanker Owners (Intertanko), the United Kingdom, and to some extent by ITOPF (see the report of the correspondence group as presented in MEPC 57/17).

This submission opened a big chapter concerning environmental risk evaluation criteria and its uses within FSA. At the 55th session of Marine Environment protection Committee (MEPC) that took place in 2006, the IMO decided to act on the subject of environmental criteria. At the 56th session of MEPC (July 2007) a correspondence group (CG), coordinated by Professor Psaraftis (the supervisor of the current work) on behalf of Greece, was tasked to look into all related matters, with a view to establishing environmental risk evaluation criteria within Formal Safety Assessment (FSA). FSA is the major risk assessment tool that is being used for policy-making within the IMO. An issue of primary importance was found to be the relationship between spill volume and spill cost.

4.5 Introduction to non linear valuation of oil spills by using compensation data

As discussed above one way to estimate the cost of oil spills can be by assuming that the total cost of an oil spill can be approximated by the compensation eventually paid to claimants. For example, such compensation information is reported by the International Oil Pollution Compensation Funds (IOPCF).

Among the first analyses was one that was performed by the IOPC Fund itself and presented in Grey(1999). 68 compensation cases were assessed mainly in order to test the limits of the compensation system. Four recent cases where IOPCF data was analyzed were known to the authors prior to their own analysis. It is not our purpose to comment on these in detail here.

Friis-Hansen and Ditlevsen (2003) used the 1999 Annual Report (except those accidents that belonged to the categories “loading/unloading”, “mishandling of cargo”, and “unknown reason” which were removed from their analysis) and converted all amounts into Special Drawing Units (SDR) by an average annual exchange rate taken from the International Financial Yearbook. Then, historic national interest rates for Money Market Rates were applied to capitalize all costs into year 2000 units followed by a conversion into 2000 USD.

Hendrickx (2007) performed an analysis based on data of the 2003 Annual Report and analyzed 91 cases by converting each compensation amount into US Dollars using for each accident the exchange rate on Dec. 31 of the year of occurrence. Exchange rates of the Bank of England were used for the currencies available and for the others an online website (OANDA.com) was used. There is no report that an inflation rate was used to bring these amounts into current Dollars.

Yamada (2009) performed a regression analysis of the amount spilled (W) and the total cost by using the exchange rate provided in the Annual Report itself. These rates can be used for conversion of one currency into another as of Dec. 31, 2007 and do not take into account the time of the accident. Furthermore, no inflation rate was used to capitalize the costs into 2008 dollars. Note, that spills less than 1 tonne were excluded by the analysis. His analysis formed the basis of Japan’s submissions to the MEPC and, to a large extent, the basis of the MEPC decision to recommend a volume-based approach.

Last but not least, Psarros et al. (2009) used combined data from two datasets, namely the IOPCF report and the accident database developed by EU research project SAFECO

II, and thus performed a regression analysis in 183 oil spill incidents. It is not immediately clear from their analysis what the SAFECO II database is and what (if any) biases it introduces to the analysis. The amounts were converted into 2008 US Dollars taking into account the inflation rate. Comments on the last two papers can be found below.

4.6 Analyses based on the IOPCF Data

Compensation for oil pollution caused by tankers is governed by four international conventions: the 1969 and the 1992 International Convention on Civil Liability for Oil Pollution Damage (“CLC 1969” and “CLC 1992”) and the 1971 and 1992 conventions on the Establishment of an International fund for Compensation for Oil Pollution Damage (“1971 Fund” and “1992 Fund”). These conventions together create an international system where reasonable costs of cleanup and damages are met, first by the individual tanker owner up to the relevant CLC limit through a compulsory insurance and then by the international IOPC Funds, if the amounts claimed exceed the CLC limits. More on compensation for oil pollution damage can be found in Jacobsson (2007), ITOPF(2010) and Liu et al. (2009). The IOPCF Annual report (2008) presents the claims that the IOPC Fund dealt with in the past. This report includes 107 accidents that are covered by the 1971 Fund and 33 by the 1992 Fund. For each accident the time and the place of accident are known and for most of the cases the volume of oil spill, as well as, the costs claimed and eventually covered by the Fund are recorded. It should be noted that the IOPCF spill database does not include US spills, as the United States is not a signatory to the above conventions.

Damages are grouped into the following categories:

- Cleanup
- Preventive measures
- Fishery-related
- Tourism-related
- Farming-related
- Other loss of income
- Other damage to property
- Environmental damage/studies

Table 4-3 presents an excerpt of the IOPCF 2008 Annual Report. Where claims are shown in the table as “settled” this means that the amounts have been agreed with the claimants, but not necessarily that the claims have been paid or paid in full. In our analysis we refer to cleanup cost as the cost that has been agreed (excluding cases where claims are pending) for clean-up of the damage and to total cost as the sum of

all costs that are presented in the report. As one may notice, there are cases where clean-up cost is the only category that appears and, thus, the total cost is equal to the cleanup cost (see for example Table 4-1, cases 2 and 4).

Table 4-3: Excerpt of the IOPC 2008 Annual Report. Adopted from IOPC(2008).

Ref	Ship	Date of incident	Place of incident	Flag State of ship	Gross tonnage (GRT)	Limit of shipowner's liability under 1969 CLC	Cause of incident	Quantity of oil spilled (tonnes)	Compensation (amounts paid by 1971 Fund, unless indicated to the contrary)
1	<i>Irving Whale</i>	7.9.70	Gulf of St Lawrence, Canada	Canada	2 261	Unknown	Sinking	Unknown	
2	<i>Antonio Gramsci</i>	27.2.79	Ventspils, USSR	USSR	27 694	Rbls 2 431 584	Grounding	5 500	Clean-up SKr95 707 157
3	<i>Miya Maru N8</i>	22.3.79	Bisan Seto, Japan	Japan	997	¥37 710 340	Collision	540	Clean-up Fishery-related Indemnification ¥108 589 104 ¥31 521 478 ¥9 427 585 ¥149 538 167
4	<i>Tarpenbek</i>	21.6.79	Selsey Bill, United Kingdom	Federal Republic of Germany	999	£64 356	Collision	Unknown	Clean-up £363 550

Before describing our analysis, it is important to comment on the limitations of the IOPC Fund dataset. First of all, it should point out that costs that IOPCF reports to the public are not ‘real’ oil spill costs. They only refer to the amount of money that was agreed to compensate the claimants. Although the IOPCF compensation figures are real and cannot be disputed, a question is if compensation figures can be taken to reasonably approximate real spill costs, or, failing that, if they can be used as realistic ‘surrogates’ of these costs.

Estimates of damages calculated by applying economic valuation methodologies, claims for compensation and the compensation eventually paid to claimants can never be equal (Thébaud et al, 2005). Furthermore, IOPCF consists of three intergovernmental organizations (the 1971Fund, the 1992Fund and the Supplementary Fund) which provide compensation for oil pollution damage resulting from spills of persistent oil from tankers only. In addition, it is further noted that admissible claims cannot be paid in full, especially in the case of large spills, since the total compensation paid is limited by the 1992 Civil Liability Convention (CLC) and the 1992 Fund to a maximum of 203 million Special Drawing Units (SDR), this is approximately US\$327 million (as at April 2008). For example, in the case of ‘Prestige’ totally 172 million Euros were paid from the 1992 Fund and CLC (IOPCF, 2009) which is only 2% of the total long term oil spill costs (Liu and Wirtz, 2006). To be more accurate, limits depend on the gross tonnage of the ship- more information can be found in IOPCF (2009).

As said before, the United States is not part of the IOPCF, which as of November 2009 numbers 103 states. The same is true of China (not including Hong Kong). Therefore, spills like the ‘Exxon Valdez’ are not included in the analysis. Furthermore, as of

November 2009, only 24 States are part of the Supplementary Fund Protocol which increased the maximum payable compensation to approximately USD 1,210 million (based on the conversion rate of the SDR to USD in April 2008). Interestingly enough, the most expensive claims (in total unit cost) come from Japan (see Table 2) which is the major contributor of the IOPC Fund and are small spills caused by mishandling of oil supply. Note that some of the spills given in Table 2 are removed from the final analysis as outliers and that in relevant studies such as the work of Friis-Hansen and Ditlevsen all spills caused by mishandling of oil supply were not taken into account.

Table 4-4: List of the most expensive spills in terms of total per tonne cost.

	Ship name	Year	Spill Size(tn)	Place	Flag	Cause
1	Plate Princess	1997	3.2	Venezuela	Malta	Overflow during loading operation
2	Daiwa Maru No 18	1997	1.0	Japan	Japan	Mishandling of oil supply
3	Shinryu Maru No 8	1995	0.5	Japan	Japan	Mishandling of oil supply
4	Volgoneft 139	2007	1,600.0	Strait of Kerch	Russia	Breaking
5	Dainichi Maru No 5	1989	0.2	Japan	Japan	Mishandling of cargo
6	Kriti Sea	1996	30.0	Greece	Greece	Mishandling of supply
7	Tsubame Maru No 31	1997	0.6	Japan	Japan	Overflow during loading operation
8	Shosei Maru	2006	60.0	Japan	Japan	Collision
9	Iliad	1993	200.0	Greece	Greece	Grounding
10	Sambo No 11	1993	4.0	Korea	Korea	Grounding

Finally, another major issue raised by many researchers is that the IOPCF claims probably underestimate the cost of oil spills since they do not include environmental damage costs. Only admissible claims are taken into account to be compensated and, practically, according to historical data, fewer than 1% contained natural resource damage assessments (Helton and Penn, 1999). Not to mention that, according to IOPC Fund, *“compensation for environmental damage (other than economic loss resulting from impairment of the environment) is restricted to costs for reasonable measures to reinstate the contaminated environment and, therefore, claims for damage to the ecosystem are not admissible”*.

The seminal paper from Helton and Penn (1999) is among the best sources of costs related to Natural Resource Damage (NRD). NRD assessments are performed in the United States during the last decades and are the best source to estimate the

environmental damage of the oil spills. The cost data concern 48 spill incidents across the US between 1984 and 1997 and according to the authors are skewed towards larger spills. Complete data are available for 30 cases and include oil spills from facilities and pipelines and even if this dataset cannot offer reliable results one of the main findings of Helton and Penn (1999) is that *“contrary to the public perception, costs for natural resource damages and assessment comprise only a small portion of total liability from an oil spill”*. NRD costs in the original dataset vary from 2.3 % (‘Arco Anchorage’) to 94.9% (‘Apex Houston’) of the total cost. It is worth to note that for the ‘Nestucca’ accident NRD cost was 20.5 % and for the most expensive in terms of total cost case in the history of US, that for ‘Exxon Valdez’ this figures comes down to 9.7%.

Taking into consideration all of the above, one might argue that IOPCF data does not represent a world-wide dataset, may not include all relevant costs and, by definition, there is an upper limit to the maximum oil spill cost that can be reimbursed. Thus, the use of such data to estimate total oil spill costs may be questioned, even in the case of oil spills caused by tankers only. On the other hand, if there are any actual costs that are paid to victims of oil pollution, this is probably as good a source to document such costs as anyone. Plus, it is clear that this analysis can be amended with additional data, to the extent such data becomes available.

In order to perform the analysis:

1. All incomplete entries and claims that were not eventually paid were removed. For example, case 4 (see Table 4-4) provides no information on the quantity of oil spilled and thus has been excluded from the analysis although the amount of clean-up cost agreed is known.
2. All claims for the cleanup and the total cost categories (in the case of multiple claims) were added up by converting them to US Dollars at the time of the accident. The author is aware of the fact that the year of the accident and the year when the amount agreed was paid are not the same but this was the only available information. Furthermore, the exchange rates used in these conversions were found in various CIA Factbooks and in a list of foreign currency units per dollar that is compiled by Antweiler (2009).
3. The cost of the previous step were capitalized into 2009 US Dollars by using conversion factors based on the Consumer Price Index (CPI).

This way we arrived at two datasets, one having data on the Cleanup Cost (CC) and the Volume (V) and another on the Total Cost (TC) and the Volume (V). These datasets were not disjoint. In fact, the first dataset contained 84 entries, the second had 91 entries, and 68 spills reported both CC and TC.

According to Friis-Hansen and Ditlevsen (2003), the logarithm of the oil spill volume and the logarithm of the total spill cost are positively correlated, having a very high correlation coefficient. This was also observed by Hendrickx (2007), Yamada (2009) and Psarros et al. (2009). Our analysis of possible fits concluded that the double logarithmic, the multiplicative and the double reciprocal have the highest correlation coefficients and R-squared values. Therefore, Costs (TC and CC) and Volumes (V) were Log-transformed and a linear regression was performed for the two cases.

The necessary conditions for a linear regression to be valid were tested and an analysis of variance (ANOVA) was also performed. Furthermore identification of outliers was performed by carefully examining studentized residuals greater than 3. The regression analysis was repeated until no outliers could be found. Finally, the linear regression formulas in double logarithmic form were transformed into non-linear regression curves. The results of the regression analyses are presented in the following section.

4.6.1 Results of the IOPCF data analysis – Cleanup Cost (CC)

After removing incomplete entries, a dataset of N=84 spills for the period 1979-2006 was used for this regression analysis (see Fig.4-5).

The minimum volume was 0.2 tonnes and the maximum was 84,000 tonnes. The average spill was 4,055.82 tonnes with a standard deviation of 14,616.15 tonnes and the median was just 162.5 tonnes. Even without a histogram one could easily realize that most claims came from relatively small spills. There were only 10 spills above 5,000 tonnes and, thus, one should be very careful when using the regression formulas to extrapolate the cost of large spills.

The equation of the fitted model using linear regression was:

$$\text{LOG}_{10}(\text{CleanupCost}) = 4.64773 + 0.643615 \text{ LOG}_{10}(V)$$

or, $\text{Cleanup Cost} = 44,435 V^{0.644}$ Eq. 4-2

The R-Squared statistic indicates that the model as fitted explains 61.5254% of the variability in LOG₁₀(Cleanup Cost). The correlation coefficient (Pearson's correlation coefficient r) equals 0.7844, indicating a strong relationship between the variables. The analysis of variances (ANOVA) indicated that there is a statistically significant relationship between LOG₁₀(Cleanup Cost) and LOG₁₀(V) at the 95.0% confidence level.

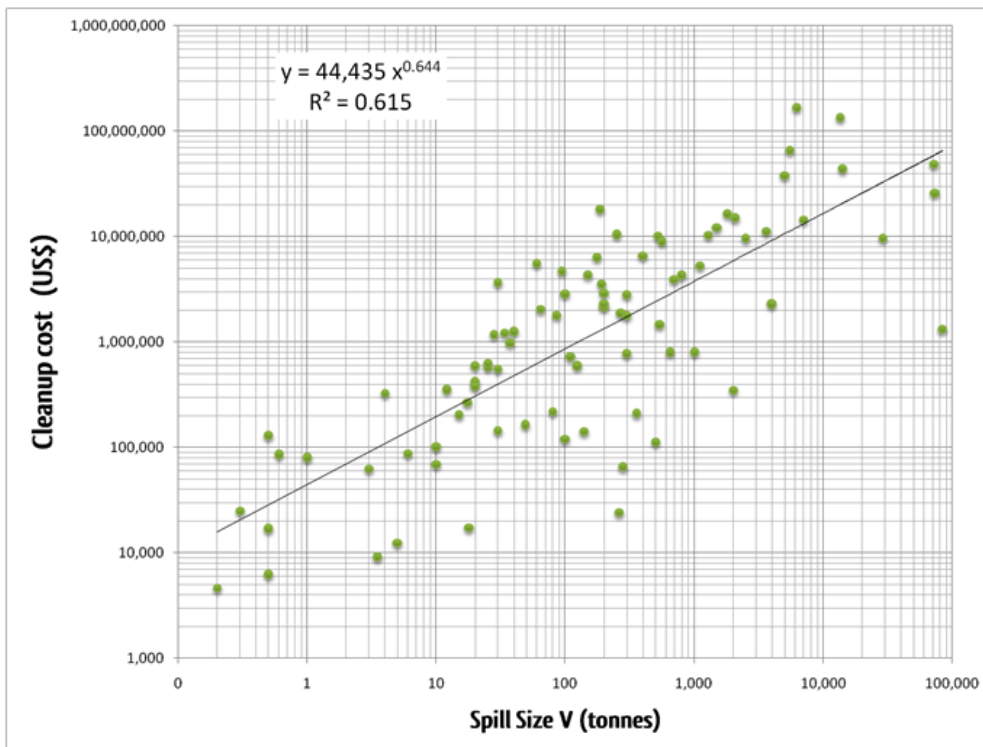


Fig 4-5: Linear Regression of Log(Spill Size) and Log (Cleanup Cost).

Furthermore, an average per tonne oil spill cleanup cost using the IOPCF database was calculated by dividing the total amount paid by the Fund for cleanup by the total amount of oil that was spilled. According to our analysis, this value came to 1,639 USD (2009) per tonne.

4.6.2 Results of the IOPCF data analysis – Total Cost (TC)

Following the same methodology as in the previous step, a regression analysis of log(Total Cost) and log(Spill Size) was performed initially for N=91 spills (for the period 1979-2006). The analysis of the studentized residuals revealed the existence of a total number of 8 possible outliers. These outliers were removed. After three consecutive regressions we arrived at the final dataset of N=83 spills (see Fig.4-6).

The minimum volume here was 0.1 tonnes and the maximum was 84,000 tonnes. The average spill was 4,854.29 tonnes, with a standard deviation of 16,064 tonnes and the median is just 140 tonnes. There are only 11 spills above 5,000 tonnes.

The equation of the fitted model using linear regression was:

$$\text{LOG}_{10}(\text{TotalCost}) = 4.71123 + 0.727567 \text{ LOG}_{10}(V)$$

or,

$$\text{Total Cost} = 51,432 V^{0.728} \text{Eq. 4-3}$$

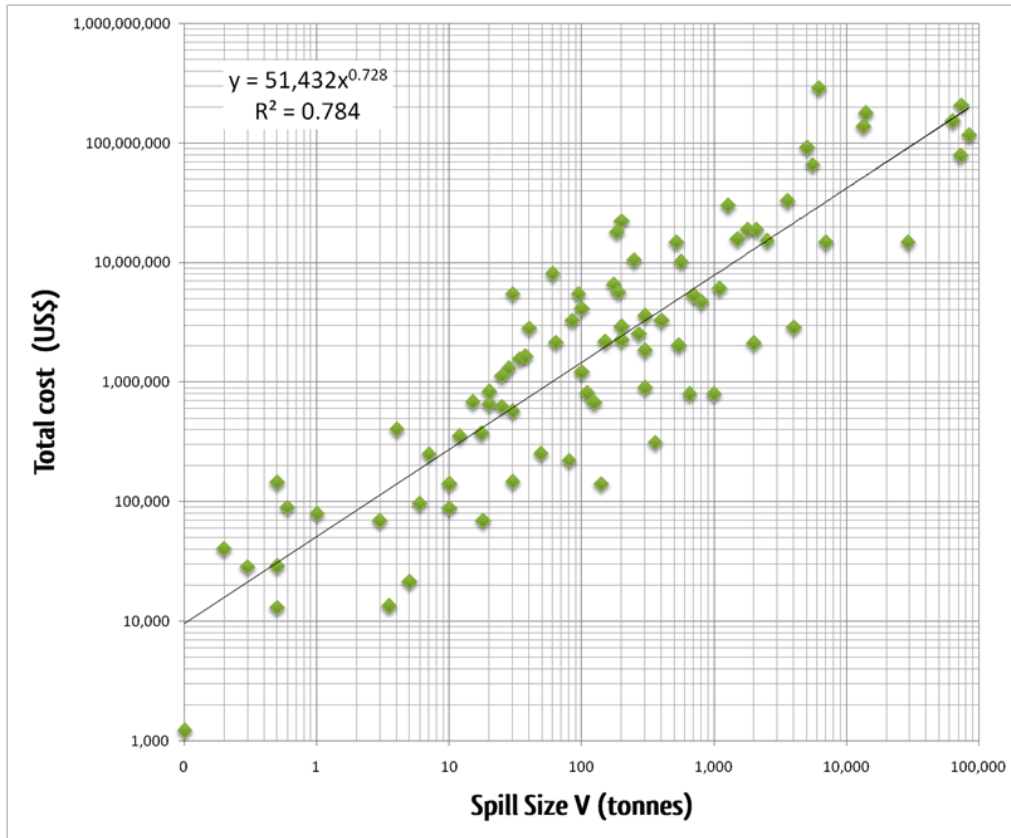


Fig. 4-6: Linear Regression of Log(Spill Size) and Log (Total Cost).

The R-Squared statistic indicated that the model as fitted explains 78.26% of the variability in LOG10(Cleanup Cost). The correlation coefficient (Pearson's correlation coefficient ρ) equals 0.8846, indicating a strong relationship between the variables. Again, the analysis of variances (ANOVA) indicates that there is a statistically significant relationship between LOG10(Total Cost) and LOG10(V) at the 95.0% confidence level.

An average per tonne oil spill total cost using the IOPCF database was also calculated by dividing the total amount paid by the Fund by the total amount of oil that was spilled. According to our analysis, this value comes to 4,118 USD (2009) per tonne.

It has to be noted that our regression analysis was very carefully performed in order to identify possible outliers given the high sensitivity of the outcome on the dataset that we chose. Outliers at both end of the spectrum were removed, that is, both for very low and for very high total spill costs per unit volume. In order to illustrate the sensitivity of including or not including such spills, we present the following for a hypothetical cost for a one tonne spill. The total cost given by the regression formula for a hypothetical oil spill of 1 tonne is 51,437 USD. The results would have changed dramatically if some outliers had not been removed. For example, let us have a look at two extreme accidents both caused by mishandling of oil supply in Japan. The 'Kifuku Maru' accident in 1982 resulted in a spillage of 32 tonnes. The amount of money

(converted into 2008 USD) that was paid for compensation was just 165 USD per tonne, a very low value. On the other hand, in 1997 the accident of 'Daiwa Maru No 18' resulted in an one tonne spillage that costed more than 4.5 million USD. If the extremely high cost value of the 'Daiwa Maru No 18' had been included in the regression the formula would produce a total per tonne cost for the hypothetical spill of one tonne of 56,058 USD. On the other hand, the extremely low, in terms of cost, case of 'Kifuku Maru' would have pushed the same value to as low as 46,706 USD.

4.6.3 Results of the data analysis – Total Cost to Cleanup Cost ratio

Vanem et al (2007a, 2007b), taking into account the work of Jean-Hansen (2003), McCay et al (2004) and Etkin (2004), concluded that a ratio of 1.5 should be assumed for the ratio of socioeconomic and environmental costs divided by cleanup costs. Thus, the total oil spill cost is 2.5 times the cost of cleanup, according to their analysis.

The data provided by the IOPCF Annual report can be used to estimate an average total cost/cleanup cost ratio, for the sample of spills for which the values of both CC and TC are available. Since we are only interested in the ratio, there is no need to do the conversions discussed before (i.e to use the exchange rate and the CPI index). Furthermore, accidents for which the claimed costs were only clean-up costs have to be removed. If cleanup cost is the only cost category available, this means that the total cost (as in the analysis performed above) would be equal to the total cost and in this case the ratio will be equal to 1. In order to remove this bias, all ratios equal to 1 have been removed, although this probably biases the analysis towards higher total cost to cleanup cost ratios. A ratio of 87,547 of the 'Braer' accident was also removed as an outlier. The dataset of the N=68 ratios that were left (see Fig. 4-5), has a minimum ratio of 1.002, a maximum of 10.01, a mean of 1.929 and a median of 1.287. The median is the measure of center (location) of a list of numbers. Unlike the mean, the median is not influenced by a few very large values in the list and may be a more appropriate criterion for this purpose.

Based on the figure below, it seems that the factor of 2.5, taken by project SAFEDOR to represent the average ratio of total spill cost to cleanup cost globally, is probably on the high side.

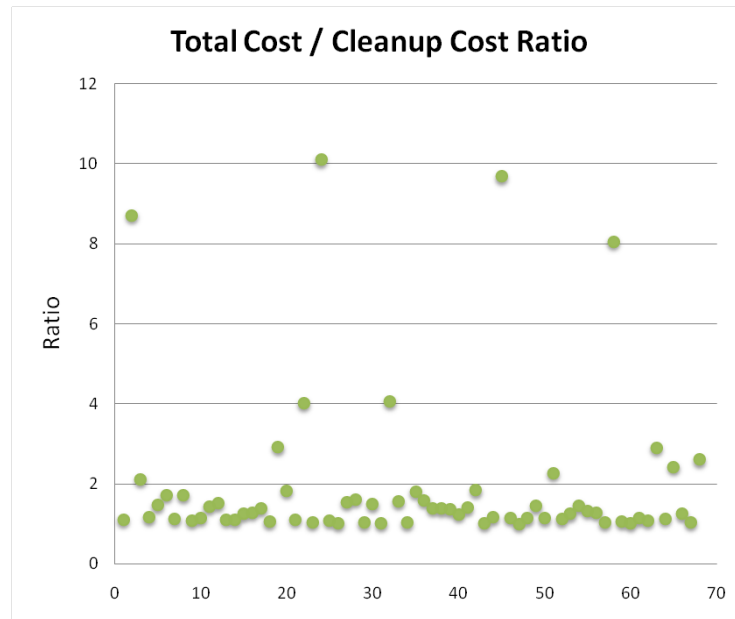


Fig. 4-7: Total Cost/Cleanup Cost Ratio. Source: Data from IOPCF(2008).

4.6.4 Comparison with similar studies

The following table summarizes the various oil spill total cost volume-based regression formulas and the corresponding R-squared values for this study, the study of Psarros et al (2009) and the study of Yamada (2009). For comparison purposes we also include the constant value of 40,000 USD/tonne for oil spill total cost as presented in Skjong et al(2005) and Vanem et al (2007a,2007b) when the authors proposed the cost effectiveness threshold value of 60,000 USD/tonne. For the 4 studies mentioned above, Figure 4-8 displays the total unit cost (in log-log plot).

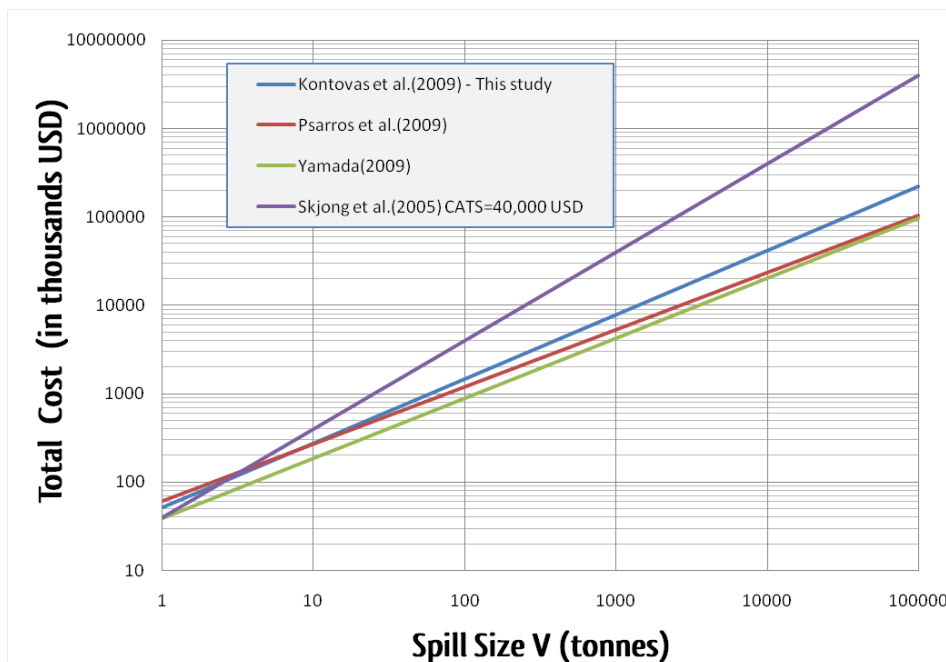


Fig. 4-7: Comparison of studies – Total Costs per tonne (in semi-log plot).

Table 4-5: Comparison of Total Cost Formulas

Study	Total Cost = f (Volume)	R ²
Kontovas et al. (2009) – this study	Total Cost = 51,432 · V ^{0.728}	0.784
Psarros et al. (2009)	Total Cost = 60,515 · V ^{0.647}	0.507
Yamada (2009)	Total Cost = 38,735 · V ^{0.66}	0.460
Skjong et al. (2005)	Total Cost = 40,000 · V	N/A

What is interesting in the above table is that our study produces a higher TC than Yamada's for all values of V, a higher TC than the one in Psarros et al.(2009) for all oil spill sizes more than about 10 tonnes, and a lower TC than in Skjong et al(2005) for all V more than about 10 tonnes. Still, Psarros et al (2009) derive a much higher average value, equal to about 54,000 USD/tonne, based on the average value of the ratio 'total cost/spill volume' for a log-normal distribution. Actually Psarros et al (2009) went one step further: they multiplied the 54,000 figure with the F=1.5 assurance factor and derived a 82,000 USD/tonne figure, which was then dropped in favor of the original constant 60,000 USD/figure. But, and for the reasons that we will outline in the next section, in this particular case we do not think that the 54,000 USD/tonne average ratio can be justifiably used in an environmental FSA.

What is equally interesting in the above table is the higher R-squared value of our study versus those of the others, implying a better fit with the data, and possibly a more reliable representation of spill costs on a volume basis. This is mainly explained by the removal of the outliers as mentioned earlier.

4.7 Analyses of compensation costs of oil spills in the US

The previous analyses do not contain oil spills from the United States since US is not a member of the IOPC Fund. However, the US submitted to the Correspondence Group raw data regarding the cleanup cost of oil spills that were covered by the *Oil Spill Liability Trust Fund (OSLTF)*. The US expressed the opinion that the data provided underestimate the real cleanup cost since "not all spills result in an opening of the OSLTF, nor are all response costs captured by OSLTF expenditures." Note that we capitalized all costs to 2009 USD prices by using the GDP deflator in line with the way that the US performed a preliminary analysis that was submitted to the CG. This was done in order to make the analysis compatible with that of the IOPCF data.

The Oil Spill Liability Trust Fund (OSLTF) was first established under Subpart C of the Omnibus Budget Reconciliation Act of 1986 (P.L. 99-509). It was later revised by the Oil Pollution Act of 1990 (OPA of 1990), revisions to the OSLTF can be found at 26 U.S.C. 9509. OPA of 1990 also consolidated requirements of prior Federal oil pollution laws (listed below) and their supporting funds.

The Oil Spill Liability Trust Fund (the "Fund") was financed by a \$.05 per barrel tax on crude oil received at refineries or on petroleum products imported to, consumed in, or warehoused in the United States, to a level of \$1 billion (26 U.S.C. 4611(f)(2)). The tax was suspended on July 1, 1993, because the un-obligated Fund balance exceeded \$1 billion (26 U.S.C. 4611(f)(2)). It was reinstated on July 1, 1994, when the balance declined below \$1 billion. The \$.05 per barrel tax expired on December 31, 1994 (26 U.S.C. 4611(f)(1)) and was reinstated by the 2005 Energy Policy Act on April 2006. In November 2008, the Energy Improvement and Extension Act of 2008 increased the tax from 5 cents per barrel to 8 cents per barrel through December 31, 2016 and to 9 cents per barrel from then until December 31, 2017.

The IOSLT dataset consists of 486 cases which are mainly extremely small spills. The median spill size of the dataset is 0.16 tonnes and the average just 168.29 tonnes.

According to a preliminary regression analysis performed by the US, results failed to demonstrate a statistically significant relation between response cost (in 2005 USD) and spill volume (in tonnes). We performed our own analysis and performed a regression analysis in the same way that was performed for the IOPCF dataset.

The following Figure presents the results of the analysis. The initial regression (dashed line) showed a very weak relationship ($R^2=0.1817$) between response (or cleanup) cost and oil spill volume. Data in red were identified as outliers and by removing them a better fit was achieved ($R^2=0.2405$). Still the results are not satisfactory. Note that no other model achieves a better result than the linear regression between the logarithms of both factors.

For the OSLTF, analysis of some empirical data from 1998-2002, which are taken from National Pollution Funds Center (2002), has been performed by Hedrickx (2007). Contrary to the IOPC Fund, the OSLTF does not deal exclusively with spills from oil tankers. In fact, between 1990 and 2002, in only around 2% of cases in which the source of the spill could be established, were oil tankers the culprits, accounting for less than 4% of total costs (NPFC (2002)). The different sources of spills may be the reason for the weak relationship between the cleanup cost and oil volume.

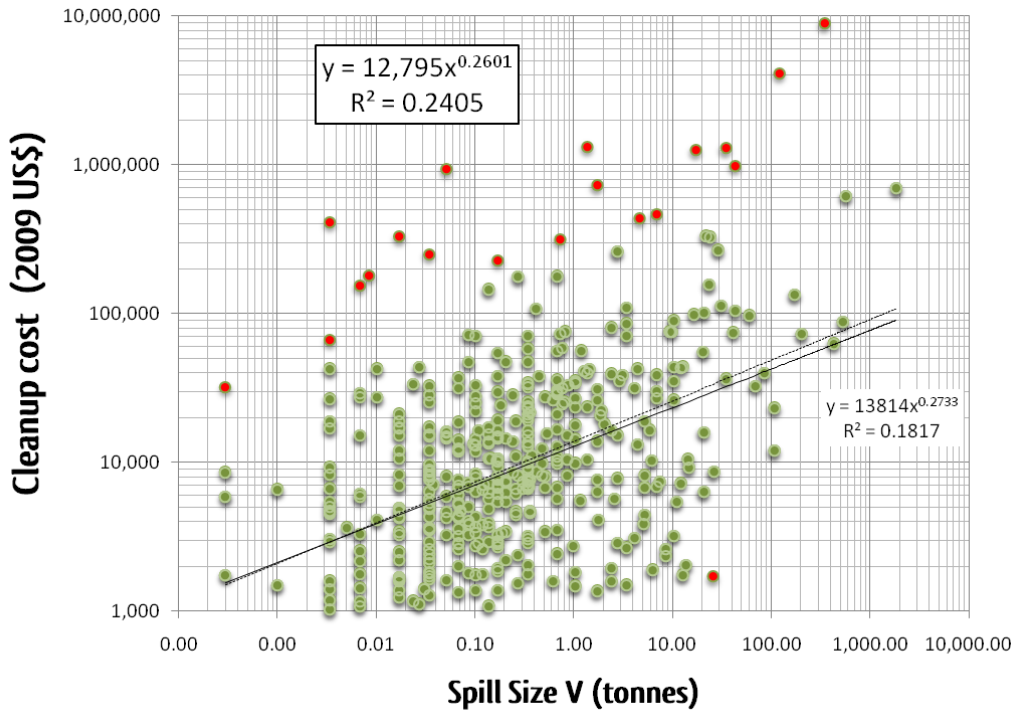


Fig. 4-8: Linear Regression of Log(Spill Size) and Log (Cleanup Cost).

Figure 4-9 presents the response cost that was actually paid by the OSLT Fund (in green). The red line represents the cleanup cost based on our regression formula (see Eq. 1) and the black dashed line is the result of the regression analysis of the OSLT data (see Eq. 4-4).

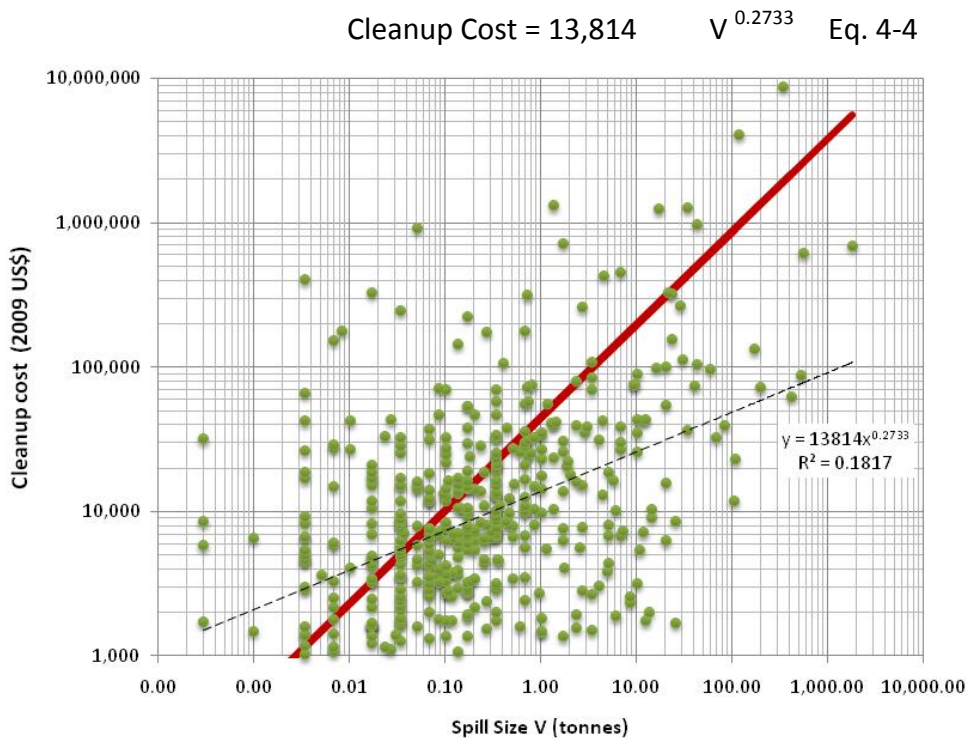


Fig. 4-9: Estimated versus real cleanup cost for spills covered by OSLT

It is therefore obvious that the nonlinear curve (as derived by the IOPCF analysis) overestimates the cost of oil spills that were covered by the OSLT Fund – that is, most of the spills (green dots) lie below the red line. In more detail, our formula overestimates the cleanup cost for 327 out of the 486 US spills (67.29%). This goes up to 80% of the cases for spills greater than 0.1 tonnes. Note again that our non-linear curve was estimated for spills higher than 0.2 tonnes.

4.8 Analysis of the combined datasets

4.8.1 Analysis of Joint IOPCF and US dataset

Although these data sets come from two different spill sources, it could be interesting to investigate the combination of the two. One reason to do so is that the IOPCF dataset includes mainly large spills and the OSLTF dataset contains mainly small spills. By combining the two datasets we arrive at a data set of 570 spills. The median oil spill has a size of 0.25 tonnes whereas the average is 749.38 tonnes.

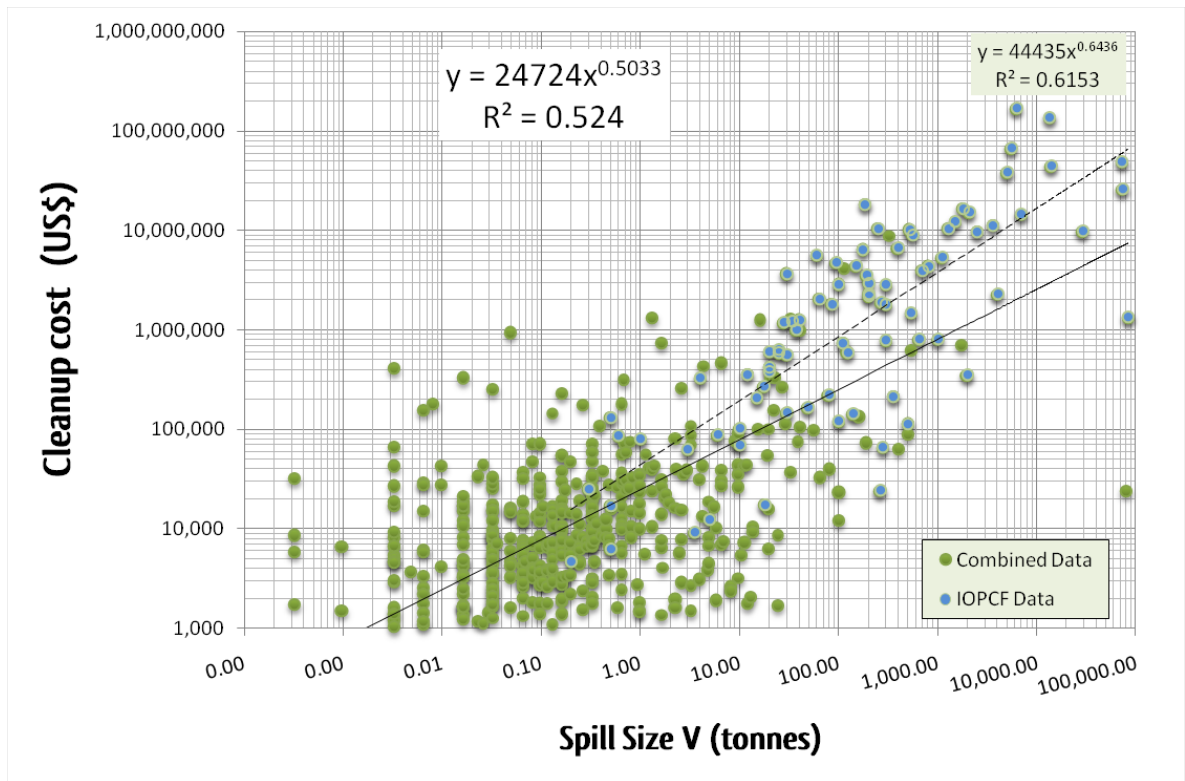


Fig. 4-10: Linear Regression of Log(Spill Size) and Log (Cleanup Cost) for combined dataset and the original regression of the IOPCF dataset (all spills included).

Data from IOPCF database (blue dots) were combined with those from OSLTF (green dots). The dashed line in Fig. 4-10 presents the line described by Eq.4-2 – that is the formula derived by the IOPCF dataset. By combining all data we arrive at a new trendline that lies well below the original, especially for big spills. The reason as

discussed before is that the OSLTF dataset contains extremely small spills and about 3 times more data than the IOPCF dataset. Note that any curve derived by regression analysis of the combined dataset will grossly underestimate spills over 1,000 tonnes.

By removing outliers (data points with an absolute studentized residual of above 2) a better fit can be achieved, see Fig. 5. The equation of the fitted model is:

$$\text{Cleanup Cost} = 24,936 V^{0.5271} \quad \text{Eq. 4-5}$$

Given that the IOPCF dataset contains spills greater than 0.2 tonnes we also performed a regression analysis of the combined dataset for spills greater than 0.1 tonnes, see Fig. 4-10.

The fit is better than the one in the previous cases ($R^2=0.68$) which is even greater than the one derived by the IOPCF dataset alone. The trendline derived by this regression is shown as the solid black one. Note that R-square is just a measure of a regression fit. By carefully looking at the scatter plot, it is obvious that this model overestimates spills at the lower end and underestimates spills at the higher end.

In any case, the basic result of our analysis is that for spills greater than 0.1 tonnes the formula that was proposed in the previous deliverable overestimates the cleanup cost even for the cases of the OSLT Fund.

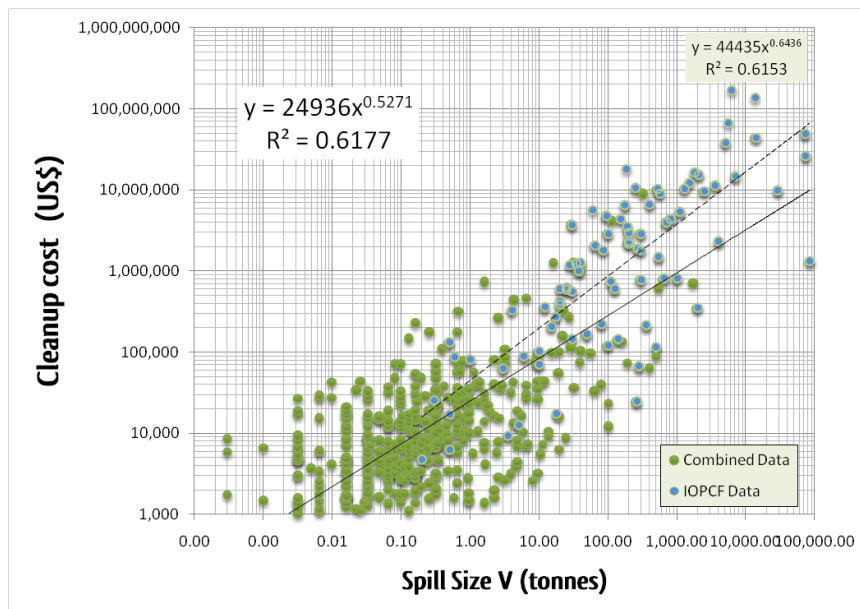


Fig. 4-11: Linear Regression of Log(Spill Size) and Log (Cleanup Cost) for combined dataset and the original regression of the IOPCF dataset (outliers excluded).

Furthermore an analysis of the combined dataset was also performed for oil spill greater than 1 tonne. The analysis of the combined data (after removing the outliers) resulted in the following equation of the fitted model using linear regression:

$$\text{Cleanup Cost} = 18,113 V^{0.6816} \quad \text{Eq. 4-6}$$

The use of this cleanup cost non-linear formula is the same as of the one derived by IOPCF curve or even by the OSLTF itself. These different formulas should be examined in detail in order to find which is more appropriate to be used.

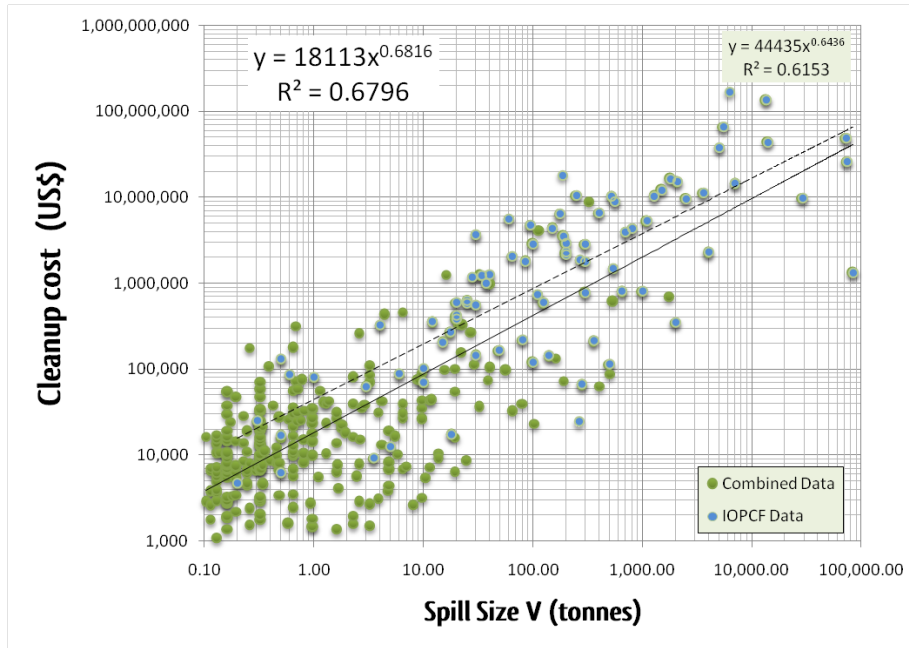


Fig. 4-12: Linear Regression of Log(Spill Size) and Log (Cleanup Cost) for combined dataset (spills greater than 0.1 tonnes – outliers excluded).

4.8.2 The Consolidated Oil Spill database presented at MEPC 62

In spring of 2011 under the initiative of Germanischer Lloyd, researchers from Japan, the US, Greece (the author being among them) and Germany developed a “consolidated oil spill database”, incorporating updated IOPCF data, data from the US and data from Norway, and performed a new set of regression analyses with a view to submit the results to MEPC 62.

Following a submission of the United States to the IMO, we had previously reported on recent analysis of oil spill cost data assembled by Oil Spill Liability Trust Fund (OSLTF), see document MEPC 61/18/2 (and the previous Section). The above dataset of US spills includes only the response cost. In order to arrive at the total cost of an oil spill a total cost to cleanup cost ratio can be used. In a submission of the US to the MEPC, see MEPC 61/INF.11, a value of \$ 40,893.64 (in 2005 USD) was given as the ‘best estimate’ of the avoided volumetric response cost and \$ 102,287.95 for the total avoidance cost.

This total cost 'best estimate' is based on literature review. Although it is out of the scope of this work to comment on the literature, the 'best estimate' for the per ton response cost (\$40,893.64) is based on the median spill size of 0.16 tonnes. The median, as well as, the average of ratios should be used with caution, see Psaraftis (2011). In our opinion, these statistics do not make too much sense. Nor does it make sense to extrapolate to large spills cost statistics derived from very small spills.

In any case, the total cost to response cost ratio for this 'best estimate' is 2.501. Therefore, based on this ratio a 'best estimate' total cost of each oil spill in the IOSLTF dataset can be estimated by multiplying the response cost by this figure. This is in line with Vanem et al. (2007a, 2007b) who taking into account the work of Jean-Hansen (2003), McCay et al (2004) and Etkin (2004), concluded that a ratio of 1.5 should be assumed for the ratio of socioeconomic and environmental costs divided by cleanup costs. Thus, the total oil spill cost is 2.5 times the cost of cleanup, according to their analysis.

In addition compensation data from 17 spills in Norwegian water is available in a document submitted by Norway to IMO, see doc. MEPC 60/17/1. These are extremely costly cases but it was decided to keep them in our analysis.

A combined dataset of these 3 sources has been jointly developed by the group of researchers after a specialized workshop hosted by Germanischer Lloyd in February 2011 and a series of discussions among the group. Various regression analyses, based on this combined dataset, were performed by the author amongst others and submitted by Greece to the IMO, see doc. MEPC 62/18/1. Greece's intent was to participate in a joint submission to MEPC 62, but this proved impossible due to lack of time in processing a joint submission. Note that minor late adjustments in the combined database produced minor differences in some relevant results that were submitted to the IMO by Germany, Japan and the United States, see doc. MEPC 62/INF.24.

Going back to the consolidated dataset, we arrived at a dataset of 357 spill with a minimum oil spill size of 0.0003, average of 12.79 and a maximum of 1,747.55. The median is just 0.16 which means that this dataset consists of extremely small spills. Therefore the combined dataset consists of 476 spills. The min spill size is 0.0003 tonnes and the maximum 144,000. The median is 0.3 and the average 1,251 tonnes. Therefore the dataset consists of very small spills. The equation of the fitted model using linear regression was:

$$\text{Total Cost (2009 USD)} = 68,779 V^{0.593}$$

Given that the IOPC Fund contains spills greater than 0.1 tonnes, regression analysis for the spills greater than 0.1 tonnes was also performed. The final dataset consists of 343 spills, having a median of just 2.27 tonnes. Obviously there is a great influence of the small spills that dominate the US dataset. The regression formula derived from spills above 0.1 tonnes of the combined dataset is the following:

$$\text{Total Cost (2009 USD)} = 42,818 V^{0.7294}$$

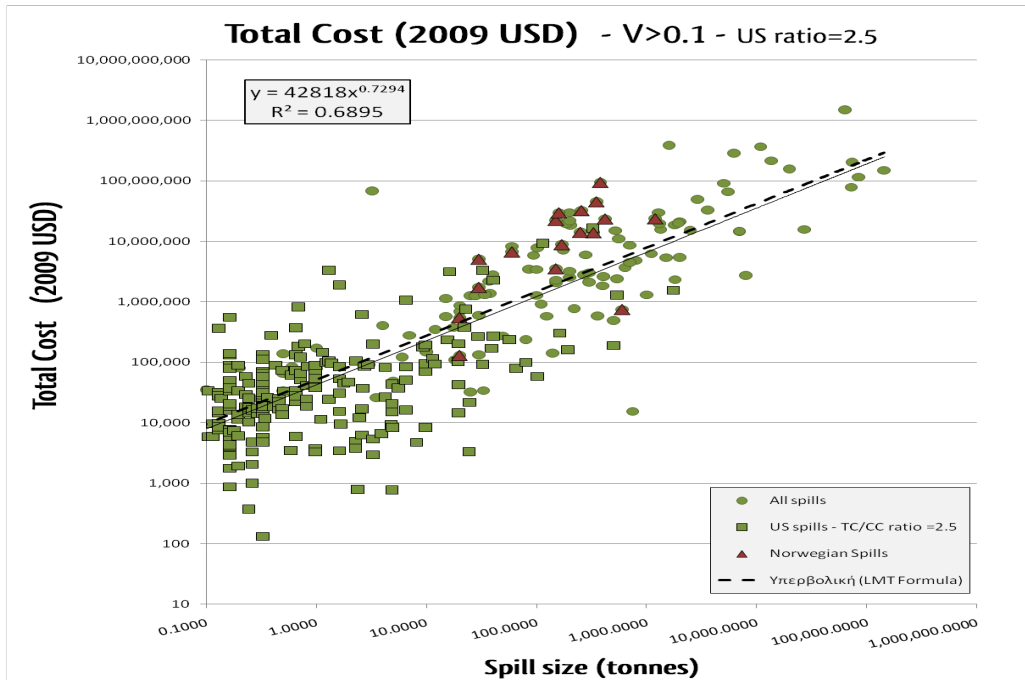


Fig 4-13: Linear Regression of Log(Spill Size) and Log (Total Cost) – US TC/RC ratio of 2.5.

Furthermore, various analyses were carried out. Regression formulas were derived for the whole dataset and subsets that contain spills greater than 0.1 tonnes (Fig.4-12), spills that happened after 1990, and spills that are over 0.1 tonnes and happened after 1990. Although it is not the purpose of this work to comment on these analyses the main conclusions are (see Fig. 4-13):

- (a) the updated formula given by the equation above (based on the consolidated dataset) underestimates the total cost in comparison to the original regression formula (Greek Total Cost Formula) for spills greater than 10 tonnes and
- (b) the original regression formula lies above all regression formulas derived from the combined dataset (at least for large spills) which means that it overestimates the total cost.

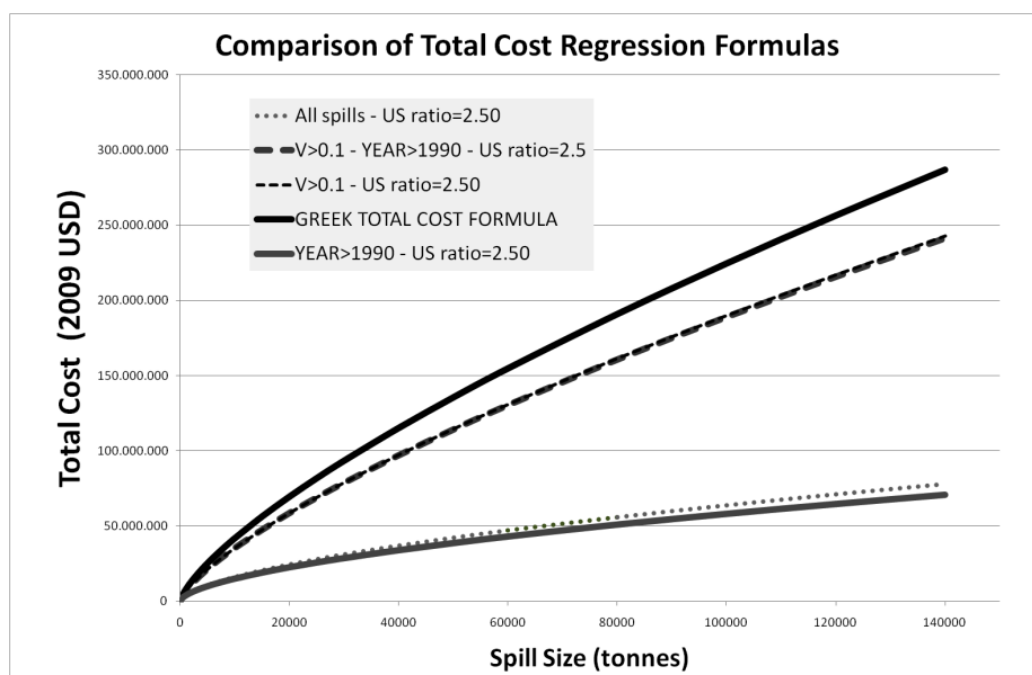


Fig. 4-13: Comparison of various regressions

4.9 Non linear cost functions within the IMO

Recall from the previous Sections that Skjong et al. (2005) and Vanem et al (2007a,2007b) presented an environmental criterion that was named CATS (for “Cost to Avert one Tonne of Spilled oil”) and its suggested threshold value was 60,000 USD/tonne. Kontovas and Psaraftis (2006) were probably the first to question the SAFEDOR approach, both on the use of any single dollar per tonne figure and on the 60,000 dollar threshold. This paper was the core of a submission by Greece on this issue (doc. MEPC 56/18/1). This submission opened a big chapter concerning environmental risk evaluation criteria and its uses within FSA. At the 55th session of Marine Environment protection Committee (MEPC) that took place in 2006, the IMO decided to act on the subject of environmental criteria. An issue of primary importance was found to be the relationship between spill volume and spill cost; most of the work presented above is part of the work performed by the author and other LMT members while working on various IMO submissions (on behalf of Greece).

Following the submission of Greece (MEPC 56/18/1), MEPC 57 (2007) noted that further work, including more research, was needed on the subject, and agreed to establish a correspondence group (CG), under the co-ordination of Greece, in order to review the draft Environmental Risk Acceptance Criteria in FSA, and submit a written report to the 57th session of MEPC. Prof. Psaraftis of NTUA, who is the supervisor of this dissertation, was assigned the task to coordinate the CG, something that lasted until 2010. After some deliberations of a Working Group, MEPC 60 reached the

conclusion, that a volume-dependent non-linear spill cost function should be used, and recommended one such function proposed by Greece for further analysis (MEPC 60/17/2); a function that was proposed by Kontovas et al (2010).

This particular function originally proposed in Kontovas et al. (2010), see Section 4.6.2, was chosen over those proposed by Norway (Psarros et al, 2010) and Japan (Yamada, 2009) as more conservative (providing higher total cost estimates for same spill size). In spring of 2011 under the initiative of Germanischer Lloyd, researchers from Japan, the US, Greece (the author being among them) and Germany developed a “consolidated oil spill database”, incorporating updated IOPCF data, data from the US and data from Norway, and performed a new set of regression analyses with a view to submit the results to MEPC 62. The regression formulas derived by this data were presented in Section 4.8.2 and formed the basis of a Greek submission to IMO. A joint submission with slightly different results was prepared by the other 3 Member-States. Perhaps the most interesting (and quite unexpected) result of the new regressions was that even though data from the US and Norway were added (and these are quite expensive spills), the new total spill cost functions obtained are *well below* the original one by Greece.

At MEPC 62 (June 2011) a Working Group (where the author was present as a member of the Greek delegation) was formed and endorsed the consolidated database although it made clear that FSA analysts are free to perform their own analysis and derive any other cost formula which they can use in FSA studies so long as it is well documented by the data. In addition, MEPC also agreed to package the main recommendations of the discussion on this topic in the form of an amendment to the FSA guidelines (see Appendix B for the full text) and forwarded the groups report (MEPC 62/WP.13) to IMO’s Maritime Safety Committee(MSC) for further action (e.g., incorporate these findings into FSA guidelines).

Note that the proposed amendments also comments on the issue of F-N diagrams within an environmental FSA by questioning the usefulness of such diagrams given the perception that the current feet is within the ALARP region and agreed that the matter should be left open until FSA studies present some results. The reader may be also referred to Sames and Hamann (2008) who proposed two ways for setting the borders of the ALARP region on an F-T diagram one of which was to assume that the current risk lever is acceptable and to Kaneko and Yamada (2010) for another way that uses the worldwide Gross Domestic Product (GDP), annual quantity of oil spilled and income by the oil transported as key parameters.

Even though certainly MEPC 62 reached an important milestone in terms of converging on a difficult topic, at the same time there are several concerns, which are out of the scope of this dissertation to analyze. Just a major one: a ratio test was again endorsed in case an RCO reduces both fatality and environmental risk. This means that CEA should be used in this case and, thus, a net cost effectiveness index where the monetized benefits from averting the oil spill will be included in the total benefits of the RCO. In addition, most experts were not convinced that such a ratio cannot be used in the case where there is no risk reduction on human health but decided to leave some space for performing CBA in this case. We feel that the problems of using a non-linear damage function (as in the case of oil pollution) in FSA would be realized the first time that this will be performed in a real FSA.

4.10 Uses of Non linear cost functions

The way that this nonlinear function that can be used to estimate the cost of an oil spill which is also the benefit of a measure that totally eliminate the risk of such an oil spills will be discussed in Chapter 6. Next Section will be brief introduction on incorporating a non linear function into Formal Safety Assessment and Section 4.10.2 will outline the way that such a function can be used to evaluate alternative tanker designs within Probabilistic Oil Outflow risk models.

4.10.1 Within Formal Safety Assessment

RCO evaluation by comparing the benefits (derived by using a function) and the costs is, in theory, presented in Psaraftis (2008) and Kontovas et al. (2010). Yamada (2009), Hammann and Loer (2010) and Yamada and Kaneko (2010) presented a way to incorporate a non-linear cost function within FSA. The latter paper forms the basis of a relevant submission to the IMO, see doc. MEPC 59/17/1 that was submitted by Japan. In reality things are a little bit more complicated as it will be shown in the following example.

This cost is not the 'real' cost of an oil spill, it is just the compensation cost of an oil spill. However these formulas are based on the best available data. Furthermore, according to the recommendations of MEPC 62 (IMO, 2011) the cost to avert an oil spill should be equal to the damage cost multiplied by two factors, namely the "assurance factor" and the "uncertainty factor". The so-called "assurance factor" is supposed to represent society's willingness to pay to prevent an oil spill instead of sustaining its damages. The "uncertainty factor" represents the fact that the compensation costs of a spill are not equal to the real costs of that spill. Taking into

account that a commonly accepted and exact estimation of the cost of averting an oil spill is not a trivial issue we assume within this paper that the damage oil spill cost is equal to the cost of averting such a spill. However, it is duly noted that this is a subject of further debate. In any case one may assume a factor to convert these into more realistic cost functions. In our analysis a factor of 1 will be used also in accordance to the Kaldor-Hicks compensation criterion where we assume that those who suffer from an oil spill can be compensated and have no further claim.

In most FSA studies an event tree is presented. For each sequence of the event tree the expected number of tonnes that will be averted is calculated as the product of the frequency of the event (P_i) and the average consequences (V_i) and is presented as $E[V]$. This is the so-called Potential Loss of cargo (PLC) value for each sequence. This value should then be multiplied with the per tonne cost (which is a function of the spill volume) to estimate the risk (denoted as $E[C_i]$) and by summing all the relevant sequences the total risk may be obtained. Another equivalent way to estimate the expected benefit of averting an oil spill by using the cost function ($Cost(V)$) is to multiplying the probability P_i with $Cost(V_i)$. These two ways lead to equivalent results. What is important to stress out is that the expected cost should be estimated before the implementation of the RCO and after it.

According to Yamada and Kaneko (2010), an RCO can be regarded as cost-effective if the following formula is satisfied

$$\Delta B - \Delta S > 0$$

where ΔB is the benefit by implementing the RCO which is the risk reduction (in monetary units) and ΔS is the cost of implementing the RCO. ΔB is the expected cost of an oil spill before the implementation of the RCO ($E[C_{org}]$) and after ($E[C_{new}]$).

Therefore, the criterion becomes

$$\Delta S < \Delta B = E[C_{org}] - E[C_{new}]$$

In practice, the discounted costs and benefits should be compared. Furthermore, in most FSA studies submitted to the IMO the event trees have not been recalculated. In most cases, the RCO was assumed to erase the risk (which means $E[C_{new}]$ equal to zero) or a risk reduction as a percent of the initial risk was estimated by expert judgment. Taking these remarks into account, applying the criterion is straightforward.

4.10.2 Probabilistic Oil Spill Risk Analysis: Environmental Performance of Tankers

In addition the Cost Benefit Analysis presented above as a part of a Formal Safety Assessment (FSA) could also be used to evaluate the environmental performance of alternative tanker designs. In general, to model the risk of an individual tanker spill, it may be argued that one has to:

1. Determine the probability of an accident ;
2. Determine the oil outflow volume given the accident as a probability distribution ;
3. Determine the spill consequence given the outflow volume by using the non linear formula.

This methodology has been applied in many studies performing oil spill risk analysis. For example, see Montewka (2009) and Montewka et al. (2010) who present the risk of collision and grounding as a random variable and uses the risk assessment process that is illustrated in the following diagram.

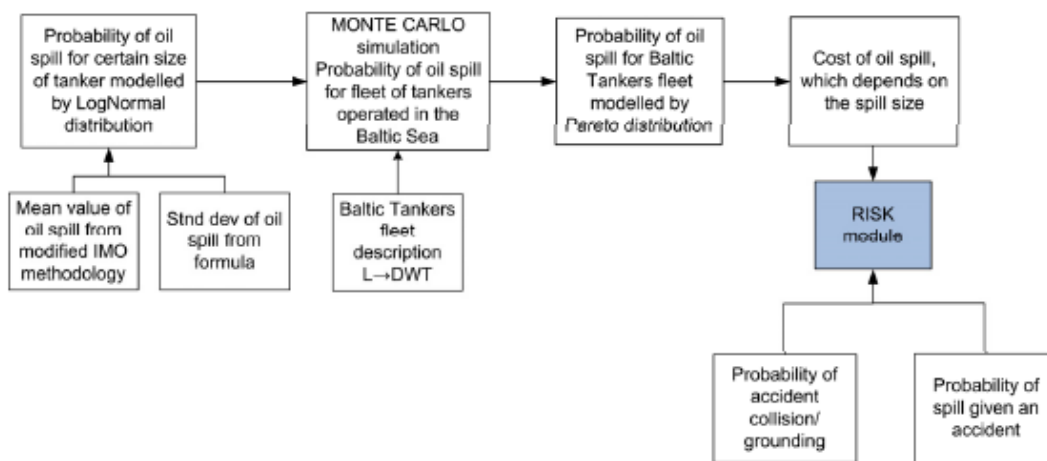


Fig. 4-14: Block diagram of risk assessment process applied in presented study [Montewka, 2010]

According to their analysis, the risk that tankers colliding or grounding pose to the environment can be calculated using the general formula, separately for collision and grounding:

$$R = P_A \cdot P_{OS|A} \cdot P_{OS} \cdot C$$

where P_A means a probability of an accident (collision or grounding), $P_{OS|A}$ means a probability of an oil spill given an accident, P_{OS} denotes a probability density function of an oil spill volume in the Gulf of Finland, C stands for consequences of an accident, which refers to an oil spill clean up costs and used the non linear function derived by Yamada (2009).

Probability density function of an oil spill volume, for the Gulf of Finland is expressed as follows (Montewka et al., 2010):

$$P_{OS} = f(x) = \frac{qb^q}{(x+b)^{q+1}}$$

where in the case of collision $q= 1.9$ and $b=9009.1$; and for grounding $q=1.5$ and $b=3847.6$ and x is a volume of spill size in tons.

Furthermore, probabilistic oil outflow models are being used in risk based optimization of crude oil carriers with respect to loss of cargo. These are in line with the IMO regulations regarding the probabilistic oil outflow for bunker tanks (applied to all spills) and cargo tanks regarding oil carriers. Indeed, MEPC has adopted a revised MARPOL Annex I/22 and 23 applicable to all new oil tankers to provide adequate protection against oil pollution in the event of grounding or collision, see IMO (2006a, 2006b, 2006c).

Regulation 22 applies to new oil tankers, which means all tankers delivered on or after 1 January 2010. The Probability density functions have been determined for the likelihood of damage being encountered at different points in the length of the ship for both side and bottom damage. An assessment is then made of the expected oil outflow from each damaged tank or group of tanks including tidal effects and accounting for any retained oil.

The mean oil outflow parameter is calculated independently for side damage and bottom damage and then combined in non dimensional value as follows:

$$O_M = (0.4O_{MS} + 0.6O_{MB}) / C,$$

where O_{MS} and O_{MB} are the mean outflows for the side damage and bottom damage respectively and C is the total volume of cargo oil in m^3 for a 98% full tank.

Thus far, research on risk based ship design has mainly focused on parametric optimization in order to reduce oil-outflow probability and increase cargo carrying capacity (Papanikolaou et al., 2010). However, the economic damage of accidental oil outflow can be estimated by using a non linear function as presented above. That way, alternative designs could be judged for their environmental performance (Sirkar et al.,1997). Ventikos & Swtiralis (2011) present a probabilistic formulation of regulation 23 of MAPPOL to calculate distribution and quantities of oil outflow for all major oil tanker categories and examine numerous cargo tank configurations for tankers by simulating multiple outflow scenarios for the tanker fleet. In addition, they perform an assessment of the cost of these potential oil spills by using some of the formulas discussed above.

5

THE ECONOMIC EFFECTS OF GHG EMISSIONS

There is a growing concern that the Earth's atmospheric composition is being altered by human activities which can lead to climate change. This view has led to the United Nations Framework Convention on Climate Change (UNFCCC) with the agreed objective *"to achieve stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system"*(see Article 2 of the Rio Convention). Due to the many uncertainties involved with climate change, the Intergovernmental Panel on Climate Change (IPCC), a scientific intergovernmental body was tasked to evaluate the risk of climate change caused by human activity. The reports produced by IPCC made it clear that differing viewpoints within the scientific community do exist. Air pollution from ships is currently at the center stage of discussion by the world shipping community. The Kyoto Protocol gave IMO -the International Maritime Organization- the task of tackling bunker emissions. Although some regulation exists for non-Green House Gases (GHGs), such as SO₂, NO_x and others, shipping has thus far escaped being included in the Kyoto global emissions reduction target for CO₂.

Developing, advocating and implementing policies is not possible without some basic understanding of the science that underlies the climate change debate and the surrounding uncertainties. Focusing on facts first, it is known that fossil fuels such as marine bunkers contain high percentage of carbon and hydrocarbons and the burning of these fuels produces carbon dioxide which is one the GHGs. Carbon dioxide enhances radiative forcing and contributes to global warming. Global Warming is the increase in the average temperature of the Earth's near-surface air and oceans. An increase beyond the normal can cause sea level to rise, decreased snow cover in the Northern Hemisphere and so on.

Schelling (2007) poses some questions that expose the relevant uncertainties. How much carbon dioxide may join the atmosphere in a business as usual scenario? How much average warming is to be expected from a specific increase in the concentration of GHGs? How will this average warming translate into climate change and what the

effects will be in 50 or 150 years from now? The stabilization of concentrations of atmospheric CO₂ will require significant reductions in global emissions of CO₂ in the future but the resultant temperature from stabilizing these concentrations at various levels (e.g., 450 ppm, 550 ppm, etc.) depends on various factors. Models estimate that the global mean surface temperature arising from a doubling of CO₂ concentrations is between 2° C and 4.5° C (IPPC, 2007).

5.1 Emissions from Shipping and Effects on Climate

As noted in the second IMO GHG Study 2009 (see Buhaug et al., 2009), transportation produces roughly 27.7% of the world's CO₂ emissions. Roughly 21.3% of those emissions are from road transportation, 2.6% from aviation, 0.5% from rail, and 3.3% from all marine transportation (2.7% comes from international maritime shipping). Carrying over 90% of world trade, and only emitting 2.7% of global anthropogenic CO₂, is a sign of a remarkably efficient industry. Shipping as the most environmentally friendly mode of transportation, serves the world trade and development and that should not be penalized by introducing non-workable solutions to its share of responsibility of Climate Change.

5.1.1 Measuring Emissions from Shipping

Total CO₂ emissions from shipping (both domestic and international) are estimated to range from 854 to 1,224 million tons (2007), with a 'consensus estimate' set at 1,019 million tons, or 3.3% of global CO₂ emissions (Buhaug, et al., 2009). However, we should note that different studies produce different results. The need for a consensus because on this subject is due to the fact that if we do not know what shipping emits we cannot set any reduction targets. Readers may recall the truism "you can't control what you can't measure". In this Section we will try to address this issue by commenting on the way emissions inventories are carried out.

Also note, in line with the above and in an effort to address the issue of GHG emissions from ships, the IMO Assembly adopted in December 2003 the Resolution A.963(23) on "IMO Policies and Practices related to the Reduction of Greenhouse Gas Emissions from Ships" which *inter alia* urges the Marine Environment Protection Committee (MEPC) to give priority to the establishment of a GHG emission baseline.

To find the equivalent CO₂ emissions, one has to multiply the bunker consumption by an appropriate emissions factor (F_{CO_2}). The 3.17 CO₂ emissions factor has been the empirical mean value most commonly used in CO₂ emissions calculations based on fuel consumption (see EMEP/CORINAIR (2002) and Endresen (2007)). The update of the

IMO 2000 study (Buhaug et al, 2008), which has been presented at MEPC 58, uses slightly lower coefficients, different for Heavy Fuel Oil and for Marine Diesel Oil. The actual values are 3.082 for Marine Diesel and Marine Gas Oils (MDO/MGO) and 3.021 for Heavy Fuel Oils (HFO). According to the report of the Working Group on Greenhouse Gas Emissions from Ships (IMO, 2008b), the group agreed that the Carbon to CO₂ conversion factors used by the IMO should correspond to the factors used by IPCC (2006 IPCC Guidelines) in order to ensure harmonization of the emissions factor used by parties under the UNFCCC and the Kyoto Protocol.

Table 5-1: Comparison of Emission Factors kg CO₂/kg Fuel (IMO, 2008b)

FUEL TYPE	GHG-WG 1/3/1	IPCC 2006 Guidelines			Revised 1996 Guidelines
		Default	Lower	Upper	
Marine diesel and marine gas oils (MDO/MGO)	3.082	3.19	3.01	3.24	3.212
Low Sulphur Fuel Oils (LSFO)	3.075	3.13	3.00	3.29	
High Sulphur Fuel Oils (HSFO)	3.021				

There are generally two main methods that can be used to produce fuel consumption and emission estimates for shipping, see Psaraftis and Kontovas (2009). One method uses marine fuel sales data in combination with fuel-related emission factors. The use of marine fuel sales to estimate emissions, also called ‘top-down’ method, or ‘fuel-based’ method, would be the most reliable method of estimating total fuel consumption and emissions if we could rely on the numbers of marine bunker fuels sales that are reported. Marine bunker supply is mainly collected from energy databases published by the Energy Information Administration (EIA), the International Energy Agency (IEA) and United Nations Framework Convention on Climate Change (UNFCCC).

However, there is widespread doubt about the reliability of bunker fuel statistics as an indicator of actual fuel used in shipping. A first problem that becomes apparent when comparing these data sets is that EIA and IEA define “International bunkers” differently: IEA gives the fuel consumption of marine international bunkers including consumption by warship, while EIA includes some international jet fuel in its figures for world fuel consumption from international bunkers. Thus, estimates of global CO₂ emissions from maritime transport derived from energy statistics differ substantially from activity-based estimates, as these are defined below.

Due to difficulties in using the top-down method, as outlined above, an alternative method has emerged. This is the so-called ‘bottom up’ method, or ‘activity-based’ method. This is an approach based on fleet activity that tries to estimate world fleet emissions by calculating emissions for all possible ship-type and size brackets. This method needs information on ship movements and ship characteristics (vessel type and size, engine type and age, fuel type, etc), as well as the corresponding fuel consumption figures and emission factors. The approach has many variants, mainly depending on how the set of inputs is obtained, and what modeling or other assumptions are used. Detailed methodologies for constructing fuel-based inventories of ship emissions have been published by Corbett and Köhler (2003), Endresen et al (2003, 2007), Eyring et al (2005), Buhaug et al. (2008) and Psaraftis and Kontovas (2009).

Table 5-2 is from Psaraftis and Kontovas (2009) and shows a comparison of bunker consumption estimates among a broad set of studies, based on the comparison made in Buhaug et al (2008). Bunker consumption is relevant as it is directly proportional to CO₂ emitted. In order to allow a proper comparison, which is difficult anyway because of different modeling assumptions, various adjustments were applied. For more information see Psaraftis and Kontovas (2009).

Table5-2 : Comparison of Bunker Consumption Results of Various Studies

	Base year	Total (Mt)	Adjusted Total	2007 est (Mt)
Eyring et al., 2005	2001	280	277	361
Corbett et al. 2003	2001	289	254	339
Endresen et al, 2007	2000	195	210	282
IMO Expert Group	2007	369	369	369
IEA total marine sales	2005	214	214	234
EIA bunker	2004	225	225	260
Buhaug et al., 2008	2007	333	333	333
Psaraftis and Kontovas, 2009	2007	298	283	283

The main reason for the different estimations among the related studies is the estimated total hours of ship engine operation per year. To be more precise, the two key parameters all ‘bottom up’ studies are the engine fuel consumption (which is based on the installed engine power (in kW), the specific fuel consumption of the engine (SFOC) and the engine load) and the engine activity – these are the engine running hours at sea, in port and laid-up. Another critical parameter that varies among studies is that of ship activity, which in many cases is derived directly from geographical ship movement data. We have noticed the use of traffic densities derived

from the International Comprehensive Ocean-Atmosphere Data Set (ICOADS) and the Automated Mutual-Assistance Vessel Rescue System (AMVER) in Endresen et al (2003) and the use of Automatic Identification System (AIS) data collected from a global network in the update 2000 IMO study (Buhaug et al, 2009). In a similar way, Corbett and Köhler (2004) based their estimation on engine profiles from manufacturer and operator survey. Of all the above and according to Buhaug et al. (2009) the number of days at sea is the parameter that contributes the largest uncertainty.

Another main difference of our approach (see Psaraftis and Kontovas (2009)) was that for most of the fleet we used real fuel consumption data as provided by ship operators. The importance of using actual data is in line with Corbett and Köhler (2004), which state that *“The importance of obtaining empirical and valid measures of vessel activity is also a point of agreement. These data are fundamental to understand engine load profiles, operating hours, and resultant fuel consumption”*.

5.1.2 Future emissions

The reader based on the above should be convinced that estimating a GHG emissions baseline is not an easy task. A fortiori, it must be much more difficult to estimate future emissions. Many scenarios for future GHG emissions, including those presented in Buhaug et al. (2009), are based on assumptions on global development in the IPCC Special Report on Emissions Scenario (SRES) storylines. Since projections of climate change depend heavily upon future human activity, climate models are run against scenarios. There are 40 different scenarios, each making different assumptions for future greenhouse gas pollution, land-use and other driving forces and divided into six categories: A1FI, A1B, A1T, A2, B1, and B2.

Buhaug et al (2009) take a similar approach as the IPCC in developing scenarios and three key categories of variables were identified. One is related to the shipping transport demand and depends on the earth's population, economic growth, modal ships etc. Another is the transport efficiency that is related to the design of the ships and the way the latter are being operated. Finally, the third category is related to cost and availability of fuels.

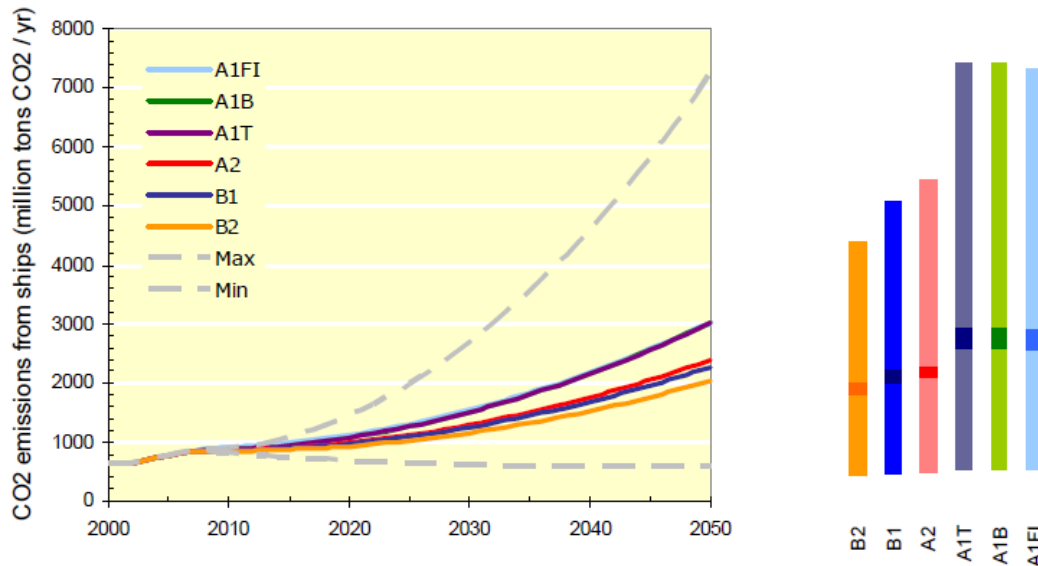


Fig. 5-1 : Trajectories of emissions from international shipping - Source: Buhaug et al. (2009)
Columns on the right-hand side indicate the range of results within individual scenario families

By carefully studying Fig.5-1 the reader may notice that extreme uncertainties exist in these trajectories. Just think about the transport demand and take the container vessels as an example. This fleet segment is extremely interesting since during the past 20 years, container transport has grown nearly 10% annually. Furthermore, according to most studies, containers are the top carbon dioxide emitters, see Buhaug et al. (2009) and Psaraftis and Kontovas (2009). However, we all witnessed a dramatic economic recession during the last two years. As a result these top polluters had to sail at lower speeds and a big percentage of the container fleet was idle. Although the models predicted some increase of the emissions, even in the short term this did not happen. Now imagine what will happen if the trans-siberian railway become a reality in 2030s and carry a share of the container traffic from East Asia to Europe. In a similar way, imagine the consequences in the case where the Arctic sea route between East Asia and Europe becomes commercially attractive. Therefore we cannot be sure of the exact harm of shipping to the environment and even if the exact amount that shipping emits and will emit in the future the true effect of shipping is not that trivial to be estimated as will be discussed in the next Section.

5.1.3 True Effect from Shipping

Regarding international shipping, it has been a fast growing sector of the global economy and its share on total anthropogenic emissions has increased lately but the nature of the contribution to climate change is complex. In addition to warming by CO₂ emissions, ship emissions of sulfur dioxide (SO₂) cause cooling through effects on atmospheric particles and clouds, while nitrogen oxides (NO_x) increase the levels of the

greenhouse gas ozone (O₃) and reduce the GHG methane (CH₄), causing warming and cooling, respectively and the result is a net global mean radiative forcing (RF) from the shipping sector that is currently strongly negative (Eyring et al.,2009). However, due to the new regulations SO₂ and NO_x, emissions will decrease and after 50 years, the net global mean effect of current emissions will be close to zero (Eyring *et al.*, 2009; Fuglestvedt *et al.*, 2008; Buhaug et al.,2009). Eyring et al. (2009), a paper co-authored by some of the same authors involved in estimation of GHG inventories and also in the second IMO study (e.g. Eyring, Corbett and Endresen) state that in 2005 the total radiative force from shipping effect measured in W/m² was -0.408, which means that currently shipping causes indeed cooling.

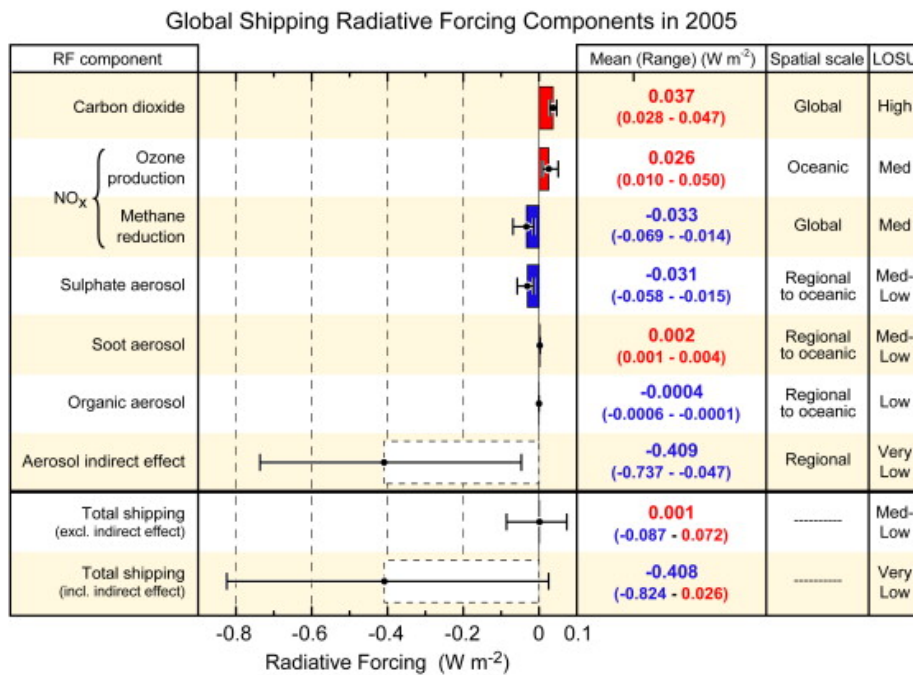


Fig. 5-2 : Shipping -Source: Eyring et al.(2009)

5.2 Emissions-related Environmental Policies in Shipping

One could go into more deep on every one of the uncertainties described above, however, the scope of this paper is to alarm the reader and not focus too much on the uncertainties. In any case, Climate Change has become a dispute and there exist divergent opinions. There is also a large number of people that are still unconvinced that man is to blame for climate change. On the other hand, the uncertainties described above should not a reason for inaction. This is in line with the so-called 'Precautionary Principle' as expressed in Principle 15 of the Rio Declaration and Article 3 of the UN Framework Convention on Climate Change (UNFCCC). Article 3 of the UNFCCC provides hence: *"The parties should take precautionary measures to anticipate, prevent or minimize the causes of climate change and mitigate its adverse effects. Where there are threats of serious or irreversible damage, lack of full scientific*

research should not be used as a reason for postponing such measures, taking into account the policies and measures to deal with climate change should be cost-effective so as to ensure global benefits at the lowest possible cost” (UNFCCC,1997).

Under this pressure, IMO has been considering a lot of possible measures including operational, technical and market-based options. From the environmental economics’ point of view, as the Stern review states, climate change is the greatest and widest-ranging market failure ever seen, presenting a unique challenge for economics. Without entering into detail, markets may fail to achieve the optimal outcome when an externality exists, that is when the actions of a firm impact on those not directly involved. That is exactly what happens in the case of air pollution. In environmental policy-making, policies are often classified in market-based, command-and-control and voluntary instruments. There are many possible ways to internalize the cost of externalities according to standard economic theory. The role of incentives, which will be briefly discussed below, such as taxes, trading schemes and command-and-control are out of the scope of this work and are being analyzed elsewhere. Note that most of the measures mentioned previously are currently being discussed at the Marine Environment Protection Committee (MEPC) of the IMO.

5.2.1 Emission Standards

There are three main types of standards: ambient, technology and emission. In brief, ambient standards are environmental quality levels in the ambient environment, such as a city or a port, and are usually expressed as average concentration level over some period of time. On the other hand, technology standards specify the technologies or techniques that should be adopted and do not specify some end result, such as a threshold level (Field and Field, 2009). For example, the requirement that all ships should be equipped with scrubbers in order to lower sulfur dioxide emissions is a technology standard. Furthermore, the regulator may specify operational measures, such as a mandatory speed reduction measure. The third type of standards are the so-called emission standards (or performance standards) and regulate the level of emissions allowed.

Emissions standards are the most popular approach to control environmental pollution and are currently being used by the IMO to control NO_x and SO_x emissions. IMO pollution rules are contained in the “International Convention on the Prevention of Pollution from Ships”, known as MARPOL 73/78. In 1997, the MARPOL Convention has been amended to include Annex VI titled “Regulations for the Prevention of Air

Pollution from Ships” which sets limits on NO_x and SO_x emissions from ship exhausts (IMO, 2008c).

Effective in July of 2005, MARPOL Annex VI (Regulation 13) set limits (in g/kWh) on emissions of nitrogen oxides (NO_x) from diesel engines that are over 130 kW. The NO_x emission limits depend on the engine maximum operating speed. Under this rule, the shipowner and, ultimately, the engine manufacturer are required to provide certification that the engine meets the IMO NO_x Technical Code when delivered to the vessel (IMO, 2008c).

The MARPOL Annex VI (Regulation 14) controls apply to SO_x emissions and include a global cap of 4.5% on the sulphur content of fuel oil. All fuel oils for use onboard a vessel covered by this annex will need to be ordered, and verified from the bunker receipt on delivery, as having a maximum sulphur content of 4.5% m/m. Furthermore, SO_x Emission Control Areas (SECAs) were established in the Baltic and Northern Sea with more stringent controls on sulphur emissions. A new emissions control area in North America will become effective in August 2012. In these areas, the sulphur content of fuel oil used onboard ships should not exceed 1.5%. The alternative is that ships must fit an exhaust gas cleaning system or use any other technological method to limit SO_x emissions to less or equal to 6 g/kWh (IMO, 2008c).

Performance standards regarding carbon dioxide emissions are also currently under discussion at the IMO and will be briefly analyzed in the following paragraphs. The IMO's Energy Efficiency Design Index (EEDI), in conjunction with the Energy Efficiency Operational Indicator (EEOI) were designed so as to help shipping achieve fuel efficiency and consequently a reduction in GHG emissions. However, the extent to which this will be truly achieved is subject to considerable debate.

Energy Efficiency Design Index (EEDI)

MEPC 58 discussed the use of the draft Interim Guidelines on the Energy Efficiency Design Index for new ships (IMO, 2008b) for calculation and trial purposes with a view to further refinement and improvement. The original objective was to establish a mandatory Energy Efficiency Design Index of the environmental performance of new ships within 2010 or 2011. At the 62nd session of IMO's Marine Environment Protection Committee (MEPC 62, July 2011), and after fierce resistance from developing countries, the IMO decided to adopt the EEDI as mandatory measure to reduce GHGs from ships (IMO, 2011).

The attained new ship Energy Efficiency Design Index is a measure of ships CO₂ efficiency and is defined as follows:

$$\frac{\left(\prod_{j=1}^M f_j \right) \left(\sum_{i=1}^{nME} C_{FMEi} SFC_{MEi} P_{MEi} \right) + P_{AE} C_{FAE} SFC_{AE} + \left(\sum_{i=1}^{nPTI} P_{PTI} - \sum_{i=1}^{nWHR} P_{WHRi} \right) C_{FAE} SFC_{AE} - \left(\sum_{i=1}^{nqff} f_{qff} P_{qff} C_{FMEi} SFC_{MEi} \right)}{fi \text{ Capacity } V_{ref} f_w}$$

The index seems complicated, but basic idea is that the numerator indicates CO₂ emission from main and auxiliary engines with a deduction from energy recovery systems that improve fuel efficiency and the denominator is based on the maximum design load condition (Capacity) and the design speed (V_{ref}).

In the case of a mandatory EEDI a baseline will be used as a limit for new designs. Given that the most obvious way to affect EEDI values is to reduce installed main engine power and thus reduce design speed, EEDI probably means a power limit for new ships.

Furthermore, deficiencies in the formula for the EEDI baseline were also identified in a submission to the IMO by Greece (IMO, 2010d) and ways on how to alleviate these deficiencies were proposed. An important caveat concerns the speed data that is used in the regressions. To the extent that ship speeds are drawn from databases, caution is necessary on how they are obtained, how they are used and how the results of the regression curves are interpreted. Furthermore, whatever the regression formula is, half of the sample ships, have an EEDI above the baseline which in and of itself, is a problem. The authors believe that there is a serious physical inconsistency between (a) the EEDI formula and (b) the formula for the EEDI baseline (the so-called EEDI ‘reference line’). In (a), and assuming that ship engine MCR grows like the cube of speed, EEDI grows like speed squared. In (b), speed does not enter the formula at all. This combination is tantamount to a speed limit, and that this speed limit can often be below the current operating speeds of several classes of ships.

Finally, Devanney (2010) states that EEDI induces owners to use smaller bore, higher RPM engines which will consume more fuel when the market is not in a boom and ships have to sail at slower speeds. In any case, EEDI is to be applied only on new designs and its potential to reduce current emissions is limited.

Energy Efficiency Operational Indicator (EEOI)

The EEOI (IMO, 2008b) is defined as the CO₂ efficiency of ships in terms of CO₂ emissions per unit transport work and may be expressed as:

$$EEOI = \frac{\sum_i FC_i \times C_{Fi}}{m_{\text{cargo}} \times D}$$

where FC is the fuel consumption, C the emission factor that converts fuel consumption to mass of CO₂ emissions, m the cargo transported and D the distance. The indicator is therefore defined as the ratio of mass CO₂ emitted per unit of transportation work, and this implies energy efficiency of a ship in operation.

The EEOI is currently voluntary and can measure the CO₂ efficiency based on its operational profile. On the other hand, EEDI is under development with the objective to arrive to a mandatory index for new designs. In practice this could also be applied to current ships but it would then be somehow similar to the EEOI. Actually, EEDI and EEOI follow the same principle, that is both indices express the ratio between the cost (i.e emissions) and the benefits that is generated (Buhaug et al., 2009).

EEOI as a performance indicator is intended as tool for evaluating the environmental efficiency of a ship or a fleet and it would allow a company to work out trends relative to the efficiency of its fleet and thus not only to achieve emission reduction but also reductions in fuel costs – that is the double dividend for a shipping company when reducing fuel consumption. Note that a mandatory EEOI has been discussed in 2008 but there is still not enough support from Member-States. In the case of a mandatory EEOI, the most effective ways to reduce the index is to steam at a lower speed or limit the cargo that the vessels are carrying. These are the easy solutions but this does not mean that these are the correct ones.

5.2.2 Market Based Measures

As regards MBMs, the discussion of which is expected to continue at the IMO in 2012, in our opinion two major classes of proposals are at the moment the most interesting: those based on a carbon ‘tax’ and those based on a trading scheme.

The proposal of a carbon tax originated in a submission to the IMO by Denmark and later supported by a submission by Cyprus, Denmark, the Marshall Islands, Nigeria and the International Parcel Tanker Association (IMO, 2010). Even though the submitters avoid the word ‘tax’ or even ‘levy’ and use the names ‘international GHG fund scheme’ or ‘contribution’ as regards what is paid, the scheme is essentially a tax on bunker fuel, paid for by the party buying the fuel. Two schemes are envisaged, one in which the tax is collected at the fuel supplier level and one in which this is done at the ship level. All ships of gross tonnage (GT) above 400 GT engaged in international trade

are envisaged to subject to this MBM. Monies collected would be allocated for purposes consistent with the objectives of the UNFCCC, such as to support mitigation and adaption activities in developing countries and to finance R&D projects on energy efficiency of ships.

The proposers argue that the IMO does have experience in setting up Funds similar to the above; see for example the International Oil Pollution Compensation Fund, a fund that compensates damages from oil spills caused by tankers. The implementation of the system also looks straightforward, as there is already a regulation for the fuel suppliers to maintain records of bunkers sold. Furthermore, according to Annex VI of MARPOL (International Convention for the Prevention of Pollution From Ships), all ships over 400 of gross registered tonnage (GRT) are required to keep details of fuel oil delivered on board for combustion purposes by means of a 'bunker delivery note' which should be retained on board for three years after the day of delivery. Among other details, the sulphur content of the fuel is recorded, as part of the SO_x emissions regulations.

On the other hand, there are some key points that have to be looked at very carefully. First of all, a more detailed regulation has to be developed in order to be able to identify ships that are engaged in international trade. Although it would be good for the environment to impose a tax on every ship, this may be too cumbersome from an administrative viewpoint. Secondly, a key element for the success of this proposal is that ships must buy fuel at registered fuel suppliers only and therefore Member-States should require that all fuel suppliers within their territory become register suppliers. Ship operators should also be required to purchase fuel only from these suppliers. A major advantage of this MBM is price certainty, which is important for investors, and administrative simplicity, at least as compared to an ETS. A possible drawback is uncertainty in the amount of GHGs that would be reduced. The second major MBM category comes under the label of emissions trading schemes (ETS). The first proposal to the IMO for a trading scheme appeared in a document submitted by Norway (IMO, 2008a; 2010a). Later, submissions of Germany, France and Norway (IMO, 2009) and of United Kingdom (IMO, 2010b) also supported this scheme. Emission reductions can be achieved by setting a cap on emissions from international shipping, and then allowing trading of emission allowances so that the target could be met. Ships will then need to surrender purchased allowances for the emissions they produce by acquiring allowances and credits from within the sector or buy them from other sectors. The allowances could be distributed by free allocation or auctioning. Auctioning is suggested due to the complexities of free allocation in the shipping sector and due to the experiences made in the EU ETS (IMO, 2009).

The establishment of a cap and target periods are crucial (IMO, 2008; 2010c). The latter states that *“the current lack of reliable bunker fuel sales statistics needs to be solved”*. And it continues *“one way of acquiring the data needed for setting a cap could make all ships present emission reports for a period of several years before the introduction of the system”*. This is absolutely correct in order to have a feasible ETS in shipping, meaning that implementation of such a scheme would be delayed.

In addition, under a trading scheme, price certainty does not exist. By definition a scheme will drive the price of permits to whatever level is required to bring emissions under its cap. This increased volatility that is evident in both the US acid rain program and EU ETS can seriously affect business investments. Investors (companies and governments) are may find it hard to make investment decisions when future carbon prices are so uncertain. Such uncertainties are, by definition, lower in the case of carbon taxes where the tax is a constant figure.

Another problem in such a scheme regards administrative burden, particularly record-keeping and verification and monitoring of the emissions. The easiest way is the requirement to record technical information related to fuel consumption. These records should be not only kept for inspection but also reported to the Flag State or to any other entity that will be responsible. This could be again solved with the use of a ‘bunker delivery note’. Finally, we should note that the implementation, high enforcement and monitoring costs of such a world-wide scheme require higher costs than of the case of a simple tax, not to mention the transaction costs.

5.3 Carbon Dioxide Valuation and Definitions

There is indeed a pressure to the IMO to adopt measures to curb carbon dioxide emissions. These measures should be able to reduce emissions at a logical cost to the society, this means that cost of regulation regulations has to be less than the benefit to the society. Therefore, the biggest challenge is to monetize carbon dioxide emissions, that is, put a monetary value to them. In some industries these values are sometimes used in the framework of broader cost benefit analysis to assess whether a particular policy is expected to improve or reduce the overall welfare of society. The analysis has to take into account all relevant costs and benefits (including impacts on climate change from averting carbon dioxide emissions).

Note that throughout this paper, reference to ‘carbon price’ means the cost per unit of carbon dioxide equivalent emissions avoided or reduced. Carbon dioxide equivalents (CO₂ eq) provide a standard of measurement against which the impacts of releasing (or avoiding the release of) different greenhouse gases can be evaluated. According to

IPPC (2007) every greenhouse gas (GHG) has a Global Warming Potential (GWP), a measurement of the impact that particular gas has on 'radiative forcing'; that is, the additional heat/energy which is retained in the Earth's ecosystem through the addition of this gas to the atmosphere.

Global Warming potentials for the greenhouse gases regulated under the Kyoto Protocol under a 100 year timeframe are as follows :

Carbon dioxide has a GWP of 1

Methane has a GWP of 21

Nitrous oxide has a GWP of 310

Sulphur Hexafluoride has a GWP of 23.90

Carbon dioxide emissions represent an environmental externality and the value of carbon emissions is not easily monetized. The greatest difficulty in carbon valuation is that environmental goods have no price since they are not marketable. One of the possible ways advocated by some circles to reduce emissions is the use of market based instruments such as carbon trading. For instance, an Emissions Trading Scheme (ETS) is used in the EU as an instrument for several industries. In that sense, a market is created where units that permit the right to emit are traded and therefore by definition this creates a market price which can be used in cost effectiveness analysis. In the current paper we review schemes of CO₂ market prices as a possible way to determine the cost effectiveness of carbon emissions' reduction actions and as a way to measure the benefits from abatement measures which is also equal to the damage cost to the society from emissions. Furthermore, within this paper we elucidate the frequently used definitions of 'carbon price'.

According to the Intergovernmental Panel on Climate Change (see IPCC,2007), Carbon Dioxide (CO₂) is

“a naturally occurring gas, and a by-product of burning fossil fuels or biomass, of land-use changes and of industrial processes. It is the principal anthropogenic greenhouse gas that affects Earth's radiative balance. It is the reference gas against which other greenhouse gases are measured and therefore it has a Global Warming Potential of 1.”

Furthermore, Carbon Price is *“what has to be paid (to some public authority as a tax rate, or on some emission permit exchange) for the emission of 1 tonne of CO₂ into the atmosphere.”*

In the models and the IPCC Fourth Assessment Report (see IPCC,2007),

“the carbon price is the social cost of avoiding an additional unit of CO₂ equivalent emission. In some models it is represented by the shadow price of an additional unit of CO₂ emitted, in others by the rate of carbon tax, or the price of emission-permit allowances. It has also been used in this Report as a cut-off rate for marginal abatement costs in the assessment of economic mitigation potentials.”

According to the literature, the most important ways to price carbon emissions are through:

- Social Cost of Carbon (SCC) or Damage Cost
- Marginal Abatement Cost of Carbon (MAC) or Avoidance Cost, and
- Market Prices (e.g by using EU Emissions Trading Scheme (ETS) Futures Price)

This next Section will comment on these methods in order to elucidate the frequently used definitions. Moreover, unlike the other, more theoretical approaches, a more realistic approach will be analyzed. In the case of environmental externalities we mostly talk of non-market impacts and in order to measure environmental damages economists either indirectly link environmental resources to some market goods or even construct a hypothetical market in which people are asked to pay for these resources. However, in the case of carbon dioxide emissions the existence of a trading scheme within the European Union allows us to use the actual amounts of money that companies pay. However, we note that the exchange price varies over time and may not be the optimal one, which will lead to the failure of this market-based approach.

5.3.1 The Social Cost of Carbon (SCC)

Stern defines the Social Cost of Carbon as a measure of *“the total damage from now into the indefinite future of emitting an extra unit of GHG’s now”* (Stern, 2007) and the IPCC as *“the discounted monetized sum (e.g. expressed as a price of carbon in \$/tCO₂) of the annual net losses from impacts triggered by an additional ton of carbon emitted today. According to usage in economic theory, the social cost of carbon establishes an economically optimal price of carbon at which the associated marginal costs of mitigation would equal the marginal benefits of mitigation.”*(IPCC, 2007)

To calculate this damage cost, the atmospheric residence time of carbon dioxide must be estimated, along with an estimate of the impacts of climate change. The impact of the extra tonne must be converted to the equivalent impacts at the time when the tonne of carbon dioxide was emitted and the impacts have to be discounted over time (Yohe et al., 2007).

According to the Intergovernmental Panel on Climate Change (see 2007 IPCC report), peer-reviewed estimates of the SCC for 2005 have an average value of \$12 per tonne CO₂ (tCO₂). In addition, the European Commission assumed an average carbon price of €39/tCO₂ in its Impact Assessment of the January 2008 Proposed Climate Change and Renewable Energy Measures from 2012 to 2020 (Curtin, 2008). Furthermore, Tol (2008) considered 211 estimates of the SCC in a meta-analysis from data gathered from a total of 47 studies and calculated a mean of \$28/tCO₂ and \$15/tCO₂ for all estimates and peer reviewed estimates respectively. For a literature review of SCC estimated the reader is referred to Comhar (2008).

The wide range of estimates is explained mostly by high uncertainties in the science of climate change and especially in the potential catastrophic impacts associated with it and with different choices of discount rate. Given that SCC *“is the present discounted value of the future stream of costs resulting from today’s emission of a new unit of carbon, future costs are discounted, or reduced in value”*, therefore in general tend to be rather low. As a result, less weight is given to policies, which means that no strong actions are being recommended, see Shanton and Ackerman (2008).

5.3.2 Marginal Abatement Cost (MAC)

Another approach that avoids the high uncertainties associated with assessing the SCC is to assess the costs of avoiding emissions. These are also referred to as avoidance costs or mitigation costs and their calculation is based on a cost-effectiveness analysis that determines the least expensive cost option to achieve a required level of greenhouse gas emission reduction.

The figure below presents an example of a **marginal abatement cost (MAC)** curve published in a global study by McKinsey in 2007 (Enkvist et al, 2007). Figure 1 shows the annual abatement needed to achieve stable atmospheric greenhouse gas concentrations of 500 ppm (parts per million), 450 ppm and 400 ppm of CO₂-equivalents. For example, a global emissions reduction of 26 Gtons of CO_{2e} per year would stabilize greenhouse gas concentrations at 450 ppm of CO_{2e}, and that reduction would need all the abatement measures up to a cost of €40 per ton of CO_{2e}. It is very interesting to note that according to Figure 5-3 the most cost-effective abatement measure is building insulation. There are indeed many measures that have a negative abatement cost – that means that carry no net life cycle cost, they come free of charge. However, the lowest cost measures are mainly efficient but cannot deliver the required emissions reductions by themselves. Policy making has to move up the cost curve progressively to more expensive technologies in order to achieve stable atmospheric greenhouse gas concentrations.

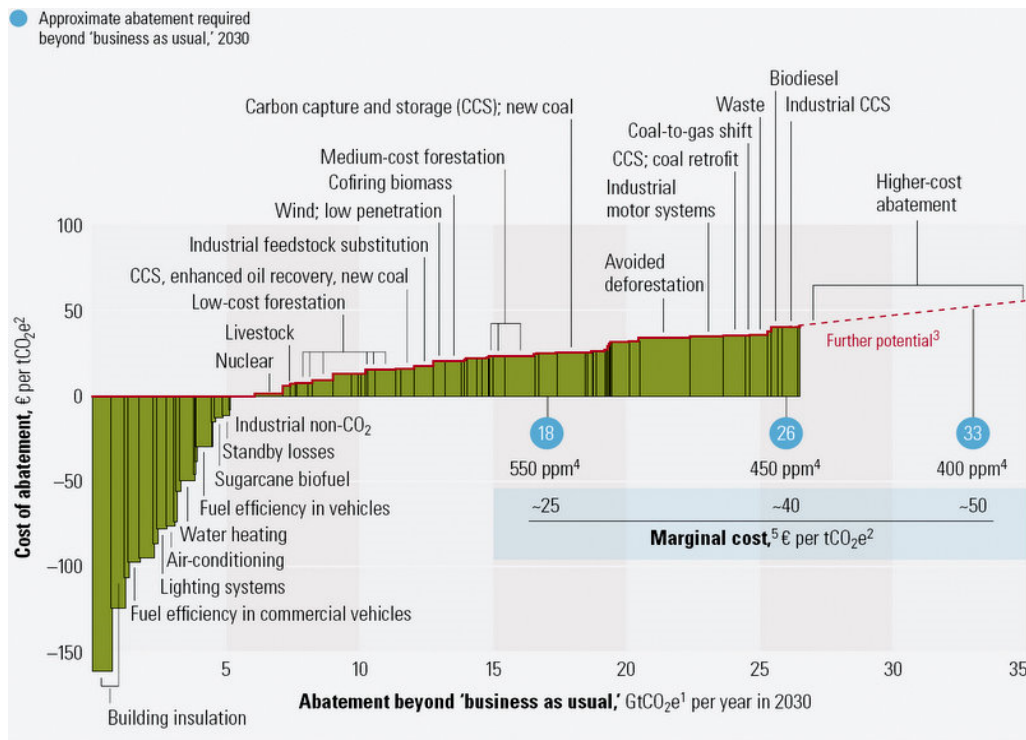


Fig.5-3: Cost curve for greenhouse gas abatement measures. Source: Enkvist et al., 2007

In a similar way, marginal abatement costs have been estimated for shipping. For example, Figure 5-3 presents the cost per tonne CO₂ averted for reduction option for the whole fleet and their potential for cutting emissions in the year 2030, see DNV(2009). Similarly curves can be constructed for fleet segments.

Figure 5-4 summarizes technical and operational measures to reduce shipping emissions from ships in 2030. The height of each bar represents the average marginal cost of avoiding a ton of CO₂ given that all measures on its left are already applied and the width represents its potential to reduce emissions. As one may notice there are several measures that have a negative cost. These seem attractive to policy makers and should be considered for mandatory implementation. Moreover, policy makers given limited budgets should adopt reduction measures according to their reduction target and the cost per measure. Furthermore speed reduction by increasing the efficiency of ports has the greatest potential to reduce emissions among all measures that have a negative cost per tonne of carbon dioxide averted and reducing speed has the greatest potential among all measures. The reader is referred to Psaraftis and Kontovas (2009) and Psaraftis et al. (2009) for more information on the effectiveness of speed reduction measures and their tradeoffs.

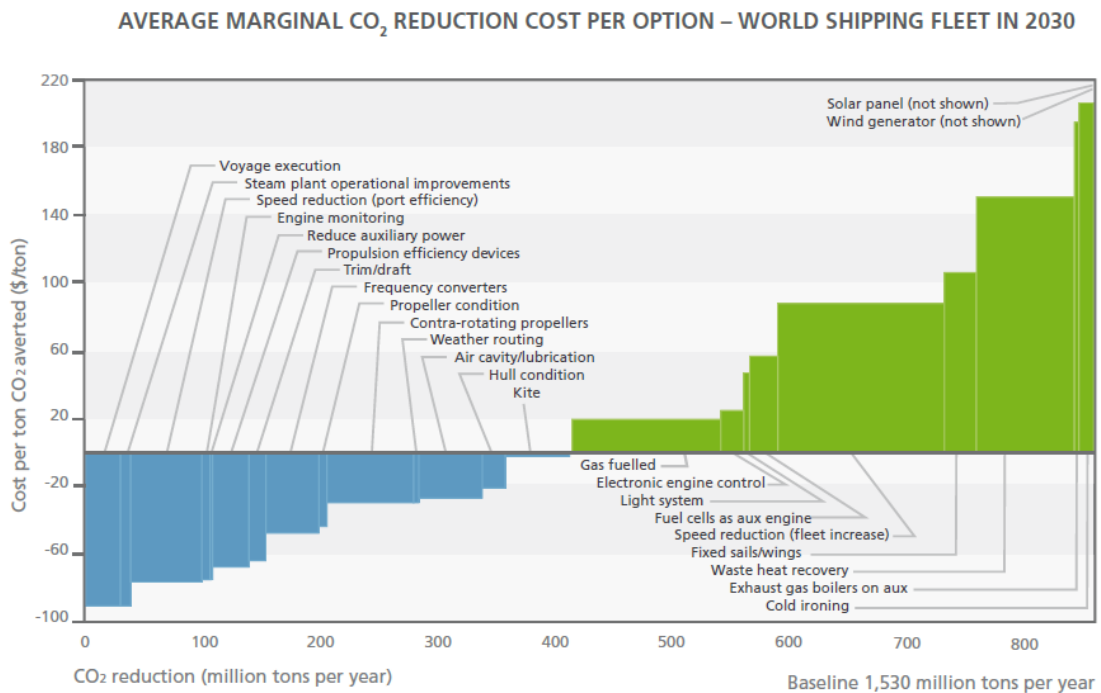


Fig.5-4: Cost per reduction option on world fleet Source: DNV, 2009

During the last years, several maritime-related Marginal Abatement Cost Curves (MACC) have been published. Faber et al. (2011a) performs a literature review of these studies.

The publicly available MACCs that focus on the shipping industry have been all submitted to the IMO and are the following:

- IMO (2009), “2nd IMO GHG Study 2009”, London.
- CE et al. (2009), “Technical support for European action to reducing Greenhouse Gas Emissions from international maritime transport”.
- DNV (2010), “Pathways to low carbon shipping” also appears as Eide et al. (2011), “Future cost scenarios for reduction of ship CO₂ emissions”.
- IMAREST (2010a), “Marginal Abatement Costs and Cost Effectiveness of Energy-Efficiency Measures”, MEPC 61/INF. 18.
- IMO(2010), “Report of Expert Group on Feasibility Study and Impact Assessment of possible Market-based Measures”, Annex 10 (MEPC 61/INF.2)
- IMAREST (2011), “Marginal Abatement Costs and Cost Effectiveness of Energy-Efficiency Measures”, update of IMAREST (2010a), MEPC 62/INF.7

Recent literature has shown that MACCs are very sensitive to numerous assumptions including the projected (up to 2020 or 2030) fuel price and fleet, the abatement measures that were included, the discount rate used in the NPV calculations, the reduction potential of each measure and the uptake of these technologies in the future. In addition, MAC estimates have some weak points. First of all, MAC prices

depend heavily on emission reduction targets and stabilization targets and, therefore, differ among sectors and countries. Furthermore, they do change massively over time as innovation kicks in with more cost-effective measures (FOEI, 2008).

Finally, note that if the carbon price derived by using MAC curves and used in policy appraisal *“is simply the most expensive measure in the strategy to meet the budgets and other overall government targets”* (f.e. emission reduction target, or stabilization levels), then any policy can be easily judged against others (FOEI, 2008). Therefore, measures that cost up to a specific threshold should be proposed for adoption.

5.3.3 Analysis of CO₂ market price

The advantage of using a market price is that it is very “real” – this figure is the one business people use when taking decisions (FOEI, 2008). This approach takes into account already existing market prices, for example the price of a carbon allowance traded in the European Union’s Emissions Trading Scheme (ETS) that gives the owner the permission to emit one tone of carbon dioxide. The main disadvantage is that the market price is based on particular sectors and on a number of countries and, thus, does not cover the whole economy.

In general, the area of Carbon Finance and of Carbon Exchanges, in particular, is a bit complicated. For a comprehensive survey of carbon exchange the reader is referred to Bettelheim and Janetos (2009). According to this survey, today there are some 20 carbon exchange initiatives, 11 of which are already trading. They are located in all geographic regions and vary from a simple matching of buyer and seller (Australian Climate Exchange) to auction markets (Asia Carbon Exchange) and from those limited to European Union Allowances (EUAs) and Kyoto Protocol Certified Emission Reductions (CERs) like the European Climate Exchange (ECX).

European Union Allowances, or EUAs, are carbon credits issued under the EU Emissions Trading Scheme to CO₂-emitting installations which is currently in Phase II (2008-2012). On the other hand, CER products ensure delivery of Certified Emission Reduction (CERs) units which are credits generated from greenhouse gas emission projects which fall under the Clean Development Mechanism (CDM) defined by Article 12 of the Kyoto Protocol.

In general EUAs are more expensive than CERs and future prices are more expensive than daily (spot) prices. The price of the EUA started to increase steadily already in the early 2005 and during the summer it peaked to over 30 Euros per tonne, see Fig. 5-5 below. However, in the end of April 2006, soon after the information regarding the true emissions EU of member states was published, the price dropped within a couple

of days from almost 30 to below 10 Euros per tonne, see POMAR/MARMET(2007) for a more detailed analysis. It is believed that the cap which was set for Phase I, was stricter than the actual emissions and therefore there was no need to buy extra allowances bringing the price eventually to zero. That is why the Phase I of ETS has been viewed as a failure.

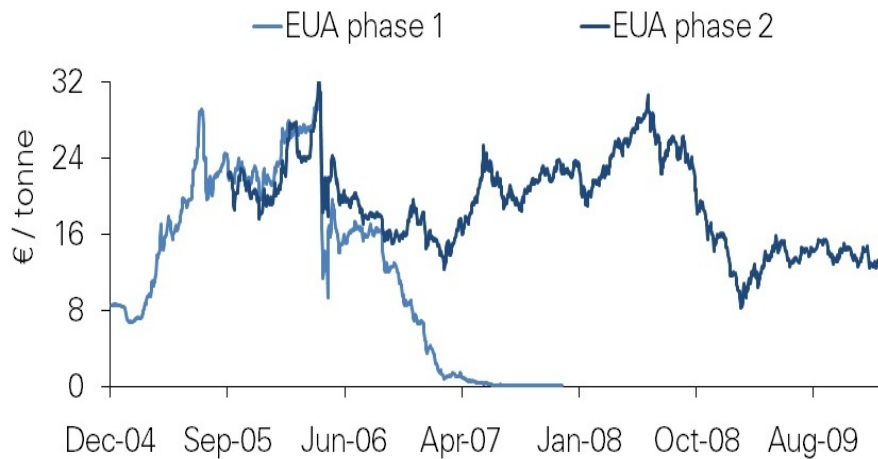


Fig. 5-5. Prices in the EU ETS, 2004-present (Front year contracts for phase 1 and 2) – Source: Point Carbon

Note that if the ETS price is applied to non-ETS sectors the result will be less emissions abatement than needed and, on the other hand, by using the same carbon price across all sectors of the economy should be perceived as fair by companies (Comhar, 2008). It looks that there is a lot of uncertainty related to carbon price and there is evidence of strong volatility in carbon markets. It is not the purpose of this paper to arrive at a single carbon price rather than present the possible ways of carbon valuation and of its uses as they will be described in the following Section.

5.3.4 The Shadow Price of Carbon

Governments in the context of policy evaluation use one, or a combination, of the above approaches to carbon valuation as the basis for incorporating carbon emissions in cost-benefit analysis and impact assessments of their regulations (DEFRA, 2007). IPCC defines as **shadow pricing** “*setting prices of goods and services that are not, or incompletely, priced by market forces or by administrative regulation, at the height of their social marginal value.*”

The ‘shadow price for carbon’, representing the cost to society of the environmental damage, has been agreed in the United Kingdom. According to the Department for Environment, Food and Rural Affairs (DEFRA,2007), the Shadow Price of Carbon (SPC) derived by estimating the Social Cost of Carbon should be used in CBA. According to DEFRA(2007) the difference between the Shadow Price of Carbon (SPC) and the Social Cost of Carbon(SCC) is the following :

“The SCC is determined purely by our understanding of the damage caused and the way we value it, the SPC can adjust to reflect the policy and technological environment. This makes the SPC a more versatile concept in making sure that policy decisions across a range of government programmes are compatible with the Government’s climate change goals and commitments.”

The Shadow Price of Carbon was set at £25.50 a carbon tonne for 2007, rising annually 2 per cent to account for the rising marginal damage cost over time DEFRA (2007). Recently, the approach to carbon valuation in UK has undergone a major review, concluded in July 2009 to a MAC-based or “target-consistent” price of carbon that involves setting a value of carbon that is consistent with the level of marginal abatement costs required to reach the targets that the UK has adopted – either at a UK, EU or international level (see DECC,2009). According to the UK Department of Energy and Climate Change (DECC) this approach *“enables more accurate policy appraisal, with respect to the costs and benefits to the UK, and consistency with our European obligations”*. For appraising policies that affect emissions in sectors covered by the EU ETS the ‘traded price of carbon’ of 21 £/tCO₂eq in 2009 values is recommended whereas for policies that affect emissions in sectors not covered by the EU ETS (the non-traded sector) the ‘non-traded price of carbon’ of 51 £/tCO₂eq should be used (DECC,2009).

In France, the Conseil d’Analyse Stratégique recommends a figure of €32/t CO₂, reaching €100 in 2030 is recommended. This value is based on a price of 27 euros proposed in the so-called 2000 Boiteux report, with inflation taken into account. This price is currently adopted to evaluate the profitability of public investments. (Centre d’Analyse Stratégique, 2008).

In the United States, the Environmental Protection Agency (EPA) and the National Highway Traffic Safety Administration (NHTSA) have also assigned a dollar value to reductions in CO₂ emissions, see US Federal Register (2010). In a joint proposal to establish a National Program consisting of new standards for light-duty vehicles that will reduce greenhouse gas emissions and improve fuel economy. benefits of reducing GHG emissions have been estimated using a set of interim global SCC values. The SCC values vary from \$34 to \$5 as these represent the estimates associated with the 3% and 5% discount rates, respectively. According to the same document, the average global SCC value used by EPA is \$20 per metric ton in 2007 dollars, a value that is recommended to be increased annually using a 3% growth rate.

Moreover, on March 9, 2010, the Department of Energy (DoE) published in the Federal Register a new final rule on energy conservation standards for small electric motors. The new rule's regulatory impact analysis includes a discussion of the social cost of carbon, which is based on a document attached as Appendix 15A to this rule. This document (EPA, 2010) is the result of a federal interagency consultative process involving, among others, the Environmental Protection Agency (EPA), the Departments of Energy (DoE), Transportation (DoT) and the Office of Management and Budget (OMB). The report (EPA, 2010) presents the 'social cost of carbon' estimates to allow agencies to incorporate the social benefits of reducing carbon dioxide (CO₂) emissions into cost-benefit analyses of regulatory actions that have small, or marginal impacts on cumulative global emissions. The interagency group selected four SCC estimates for use in regulatory analyses. For 2010, these estimates are \$5, \$21, \$35, and \$65 (in 2007 dollars). The first three estimates are based on the average SCC across models and socio-economic and emissions scenarios at the 5, 3, and 2.5 percent discount rates, respectively and the fourth value is included to represent the higher than-expected impacts from temperature change. The central value is the average SCC across models at the 3 percent discount rate. Therefore it is expected that many future analyses will use the price of \$21 per tonne CO₂.

5.4 Cost Effectiveness Index to access CO₂ emissions

The criterion that is applied in CEA is the Cost Effectiveness Index (CEI). Based on the indices defined in the FSA framework we can define the following two indices that can be used when assessing air emissions from shipping.

Gross Cost Effectiveness Index (GCEI)

$$GCEI = \frac{\Delta C}{\Delta R} \quad \text{Eq. 5-1}$$

Net Cost Effectiveness Index (NCEI)

$$NCEI = \frac{\Delta C - \Delta B}{\Delta R} \quad \text{Eq. 5-2}$$

where

ΔC is the cost per ship of the action (measure, RCO etc) under consideration [\$]

ΔB is the economic benefit per ship resulting from the implementation [\$]

ΔR is the change in mass of emissions averted per ship.

The mass of emissions averted can be 'tonnes of CO₂', or 'tonnes of NO_x' or any other exhaust gas. Particularly, for the case of Greenhouse Gases (GHG) the notion of CO₂-eq can be used.

The criterion in this case is that actions with cost effectiveness index that lie below a specific threshold, referred to as lambda (λ) value, are deemed to be cost effective and

should be adopted. We should finally note that Eide et al. (2009) have defined a net cost effectiveness index to be used within Formal Safety Assessment. According to their work, CATCH (Cost of Averting a Tonne of CO₂-eq heating) *“is a measure of cost-effectiveness in terms of the present value of the sum of the discounted current and future benefits and costs arising from implementing a given proposal at the ship level”*. The relative threshold is estimated to be 50 \$ per tonne and is the amount that shipping is expected to pay per tonne of carbon dioxide abated to *“help bridge the gap between the expected global baseline emissions in 2030 and the target emissions needed in order to limit temperature increase of 2 degrees C”*. As one can understand this threshold is based on the high uncertainties involved with the mechanisms, costs and benefits to achieve the “2 degrees C guardrail”, therefore this value may not be appropriate and has to be further analyzed.

5.5 Valuation of GHG emissions

Following the definition of the IPCC Fourth Assessment Report (see IPCC, 2007) that we presented above, the carbon price represents the shadow price of an additional unit of CO₂ emitted and it has also been used as a cut-off rate for marginal abatement costs in the assessment of economic mitigation potentials.

Different values are used among various sectors and there is no universally accepted carbon price exists, as clearly demonstrated in the previous Sections. Noting that further research is need we feel that the values used in the UK (see DECC,2009) could be a good starting point. In effect, for appraising policies that affect emissions in sectors covered by the EU ETS the ‘traded price of carbon’ of 21 £/tCO₂eq in 2009 values is recommended whereas for policies that affect emissions in sectors not covered by the EU ETS (the non-traded sector) the ‘non-traded price of carbon’ of 51 £/tCO₂eq should used (DECC,2009).

Given such a per unit price (constant or non-constant) the use of the carbon price is straightforward: Suppose that the carbon price is known and is CP in monetary units divided to mass units (for example \$/tonnes). The damage of an action is the product of the amount of emissions and the carbon price. That is, if the action produces x tonnes of carbon dioxide and the carbon price that we use in our analysis is CP \$ per tonnes then the damage is $D(x) = CP(\$ / tn) \cdot x(tn)$. This is also the benefit of an action or policy that reduces emissions by x tonnes.

In Cost-benefit analysis we then compare the benefit $B(x) = CP(\$ / tn) \cdot x(tn)$ with the implementation costs of the action. This approach can be easily generalized in the case

of a non linear damage function. In that case CP will be a function of the amount of emissions produced.

Furthermore, since carbon price has also been used as a cut-off rate for marginal abatement costs in the assessment of economic mitigation potentials, this value can be used as a cost-effectiveness threshold, the lamda (λ) value that was presented in Section 2. Among actions the one that achieves the lowest cost-effectiveness is better. As discussed above there is no FSA study until now that accesses risk by carbon dioxide emission although some criteria do exist, Eide et al. (2009). Although a constant per tonne price is still debatable the fact that market prices do not depend on the amount of permits bought is a reason for using a constant per tonne value and the ones presented in Section 5.3.4 can be a good starting point. Definitely more analysis is need on this subject.

To sum up, we presented in theory the way that the carbon price can be used within CBA and CEA. Next section will present how can a constant value be used as the cost effectiveness threshold in CEA for assessing the effectiveness of speed reduction measures. The following example is based on Psaraftis and Kontovas (2009b).

5.6 An Example : Evaluating speed reduction measures

5.6.1 Algebra of Speed Reductions

At first glance, speed reduction may look like killing three birds in one stone: reduce fuel bill, reduce emissions, and, in a depressed market, help support freight rates by artificially shrinking the fleet supply curve. The question is, is this really so?

Assuming a given ship, and for speeds that are close to the original speed, the effect of speed on fuel consumption is assumed cubic, that is, the daily fuel consumption F at sea at speed V is a cubic function of speed V . The cubic law follows from hydrodynamic principles and is a standard assumption in most analyses. Note that this is only an approximation, and one that is usually valid for small changes in speed. If the speed changes drastically, for instance from 20 to 10 let alone 5 knots, one would expect a different relationship between V and F . But also in some other cases exponents higher than 3 may be applicable. Whatever the relationship is, assuming a cubic law causes no loss of generality, as our analysis would need minor modifications in case a different functional relationship is assumed.

A cubic law means that $F=kV^3$, where k is a known constant, which is a function of the loading condition of the ship and of other ship characteristics (e.g., engine, horsepower, geometry, age, etc). An implicit assumption in this analysis is that the ship's power plant would still be able to function efficiently if speed is reduced. Speed reduction usually requires reconfiguring the engine so that its operation is optimized at the reduced load. Note that the cubic law is a rough approach, see Kontovas and Psaraftis (2011) for more.

Psaraftis and Kontovas (2009b) investigated a simple logistical scenario to examine tradeoffs between ship CO₂ emissions and other attributes of the ship operation. The scenario assumed a fleet of N identical ships (N : integer), each of capacity (payload) W . Each ship loads from a port A, travels to port B with known speed V_1 , discharges at B and goes back to port A in ballast (empty), with speed V_2 . Assume speeds are expressed in km per day. The distance between A and B is known and equal to L (km) and the total time in port at both ports is T_{AB} (days). Assume these ships are chartered on a term charter and the charterer, who is the effective owner of this fleet for the duration of the charter, incurs a known operational cost of O_c per ship per year. This cost depends on market conditions at the time the charter is signed and includes the charter to the shipowner and all other non-fuel related expenses that the charterer must pay, such as canal tolls, port dues, cargo handling expenses, and so on. Not included in O_c are fuel expenses, which are also paid by the charterer, and which depend on the actual fuel consumed by the fleet of ships. The latter depends on how the fleet is used.

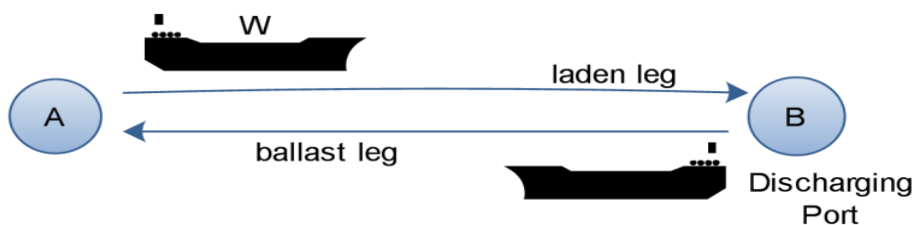


Fig. 5-6: Ship Route

Assume that each ship's operational days per year are D ($0 < D < 365$), a known input, and that the total daily fuel consumptions (including both main engine and auxiliaries) are known and are as follows for each ship: f (tonnes per day) in port, and F_1 , F_2 (tonnes per day) at sea, for the laden and ballast legs (respectively).

As stated earlier, the effect of speed change on fuel consumption is assumed cubic for the same ship, that is, $F_1=k_1V_1^3$ and $F_2=k_2V_2^3$, where k_1 and k_2 are known constants (typically $k_1 > k_2$).

In addition to the standard costs borne by the charterer, Psaraftis and Kontovas (2009b) also took into account *cargo inventory costs*. These costs are assumed equal to I_C per tonne and per day of delay, where I_C is a known constant. In computing these costs, it is assumed that cargo arrives in port ‘just-in-time’, that is, just when each ship arrives. In that sense, inventory costs accrue only when loading, transiting (laden) and discharging. These costs are called ‘in-transit inventory costs’. Generalizing to the case where inventory costs due to port storage are also considered is straightforward.

If the market price of the cargo at the destination (CIF price) is P (\$/tonne), then one day of delay in the delivery of one tonne of this cargo will inflict a loss of $PR/365$ to the cargo owner, where R is the cost of capital of the cargo owner (expressed as an annual interest rate). This loss will be in terms of lost income due to the delayed sale of the cargo. Therefore, it is straightforward to see that $I_C = PR/365$.

Based on the above, after some straightforward algebraic manipulations, Psaraftis and Kontovas (2009b) have calculated that, for N ships, total fleet costs in a year are equal to

$$D \cdot N \frac{p \left[T_{AB} f + L (k_1 V_1^2 + k_2 V_2^2) \right] + I_C W \left(T_{AB} + \frac{L}{V_1} \right)}{\frac{L}{V_1} + \frac{L}{V_2} + T_{AB}} + N \cdot O_C \quad \text{Eq.5-3}$$

Let us now assume that the speed of all ships in the fleet is reduced by a common amount $\Delta V \geq 0$. To maintain annual throughput constant, we have to add ΔN additional ships, assumed identical in design to the original N ones. It is easy to check that we can compute ΔN (which can be fractional) from the following equation:

$$\Delta N = N \left(\frac{\frac{L}{V_1 - \Delta V} + \frac{L}{V_2 - \Delta V} + T_{AB}}{\frac{L}{V_1} + \frac{L}{V_2} + T_{AB}} - 1 \right) \quad \text{Eq.5-4}$$

An implicit but very important assumption here is that these ΔN ships are readily available and can be immediately incorporated into the original fleet at a cost equal to O_C per ship per year, the same as that paid to charter the original N ships. However, a constant O_C may not necessarily be the case: (a) if there is a lack of supply of available ships, which may happen if speed reduction is applied uniformly, or for any other reason, this may have as a result a lower total fleet supply and an increase of charter rates to levels above O_C ; (b) charterers may be unwilling to pay the same cost if the speed of the vessel is lower. Charter rates will generally depend on the interaction between supply and demand, but in this model we assume them constant.

If $V_1 = V_2 = V$ (this may not mean that $k_1 = k_2$), the difference in total fleet costs is equal to:

$$\Delta(\text{total fleet costs}) = -NL\Delta V \frac{-pD(2V - \Delta V)(k_1 + k_2) + \frac{I_C WD + 2O_C}{V(V - \Delta V)}}{2\frac{L}{V} + T_{AB}} \quad \text{Eq.5-5}$$

The difference in fuel costs alone (costs after minus costs before) is equal to

$$\Delta(\text{total fuel costs}) = -NL\Delta V \frac{pD(2V - \Delta V)(k_1 + k_2)}{2\frac{L}{V} + T_{AB}} \quad \text{Eq.5-6}$$

An interesting observation is that fuel cost differentials (and, by extension, total fleet cost differentials) are independent of port fuel consumption f . This can be explained by noting that the new fleet string, even though more numerous than the previous one, will make an equal number of port calls in a year, therefore fuel burned while in port will be the same.

It is also interesting to note that for $\Delta V \geq 0$ and for all practical purposes the differential in fuel costs is always negative or zero, as difference $2V - \Delta V$ in Eq.5-5 is positive for all realistic values of the speeds and of the speed reduction. This means that speed reduction cannot result in a higher fuel bill, even though more ships will be necessary. The same is true as regards emissions, as these are directly proportional to the amount of fuel consumed.

This result can be generalized to logistical scenarios that are more complex than the one examined here, one for instance that involves a ship which visits a set of ports and is less than full (which is typically the case for containerships). The core result from this analysis is that total CO₂ emissions will be always reduced by slowing down, even though more ships would be used. The higher the speed, and the higher the speed reduction, the higher this emissions reduction will be. The theoretically maximum reduction will be if we reduce speed all the way to zero, in which case both fuel costs and emissions will also be zero. Of course, such a scenario would not make any sense, as no cargo would be moved and hence cargo inventory costs and total costs would go to infinity. By the same token, a scenario of super-slow speed may suffer from similar problems.

5.6.2 The Cost to Avert a Tonne of CO₂

How much would it cost to avert one tonne of CO₂ via speed reduction? Note here that speed reduction being only one of possible emission reduction measures, this question can be posed for any of these measures.

We address this question by noting that in expressions 5-5 and 5-6, Δ(total CO₂ emissions) equals minus total CO₂ averted by implementing a speed reduction scheme. We define as the cost to avert one tonne of CO₂ (CATC) the ratio of the total net cost of the fleet due to CO₂ speed reduction divided by the amount of CO₂ averted by speed reduction. After some algebraic manipulations, and for the simple scenario examined in the previous section, it can be shown that (Psaraftis and Kontovas, 2009b):

$$CATC = \frac{-pD(2V - \Delta V)(k_1 + k_2) + \frac{I_c WD}{V(V - \Delta V)} + \frac{2O_c}{V(V - \Delta V)}}{F_{CO_2} NDL \Delta V (2V - \Delta V)(k_1 + k_2)} \quad \text{Eq.5-7}$$

After some algebraic manipulations, this can be rewritten as

$$CATC = \frac{I_c WD + \frac{2O_c}{D}}{F_{CO_2} V(V - \Delta V)(2V - \Delta V)(k_1 + k_2)} - \frac{p}{F_{CO_2}} \quad \text{Eq.5-8}$$

The above equation obviously takes into account only CO₂ averted in the specific mode under consideration, and does not include possible CO₂ generated in other modes due to modal shifts that may be the results of such a speed reduction.

CATC is a positive linear function of both I_c and O_c and a negative linear function of the price of fuel p . It can also be seen that the denominator in the bracket is a cubic function of speed, reflecting the functional relationship between speed and the quantity of CO₂ that is produced. In addition, the last term in the above equation, $-p/F_{CO_2}$, where p is the price of one tonne of fuel and F_{CO_2} is the CO₂ emissions factor, can be recognized as the cost of the amount of fuel saved (not spent) that would produce one tonne of CO₂. This is an opportunity cost that we will have to subtract from the total cost incurred, as it corresponds to the amount of fuel that would be saved if one tonne of CO₂ is averted.

The CATC criterion can be used whenever alternative options to reduce emissions are contemplated. In that sense, the alternative that achieves the lowest CATC is to be preferred.

5.6.3 An Example: Speed reduction of a Containership Fleet

Our illustrative example investigates the effect of speed reduction in containerships. According to Psaraftis and Kontovas (2009), containerships are the top CO₂ emissions producer in the world fleet. Just the top tier category of container vessels (those of 4,400 TEU and above) are seen to produce CO₂ emissions comparable on an absolute scale to that produced by the entire crude oil tanker fleet.

Assuming a hypothetical string of N=100 (identical) Panamax containerships, each with a payload of W= 50,000 tonnes, if the base speed is V= 21 knots (both ways) and the fuel consumption at that speed is 115 tonnes/day, then for a fuel price of p= \$600/tonne (corresponding to a period of high fuel prices, before the slump of 2008), the daily fuel bill would be \$69,000 per ship. Running the same type of ship at a reduced speed V-ΔV = 20 knots (one knot down), the fuel consumption would drop to 99.34 tonnes/day (cube law vs. 21 knots) and the daily fuel bill would drop to \$59,605 per ship.

Assume these 100 ships go back and forth a distance of 2,100 miles (each way) and are 100% full in one direction and completely empty in the other. This is not necessarily a realistic operational scenario, as containerships visit many ports and as capacity utilizations are typically lower both ways, depending on the trade route. The scenario of trade routes from the Far East to Europe or from the Far East to North America, which are almost full in one direction and close to empty in the other is probably close to the assumed scenario. However, a generalization of this analysis to many ports and different capacity utilizations in each leg of the trip should be straightforward. For simplicity, assume D=365 operating days per year and zero loading and unloading times. For non-zero port times, the analysis will be more involved but will lead to similar results.

Then we will have:

AT FULL SPEED (case A):

Total fuel burned/year/ship: 115 tonnes/day*365 = 41,975 tonnes

For 100 ships = 4,197,500 tonnes

Total annual fuel cost (x\$600) = \$2,518,500,000.

AT REDUCED SPEED (case B):

To reach the same throughput of cargo per year, we will need 105 ships.

Total fuel burned/year/ship: 99.34 tonnes/day*365 = 36,259 tonnes

For 105 ships = 3,807,256 tonnes

Total annual fuel cost for 105 ships = \$2,284,353,741 (reduced vis-à-vis case A).

Thus, there is a reduction of CO₂ emissions (per year): 1,237,073 tonnes.

Fuel cost difference (per year) = \$234,146,259 for 5 more ships, that is, \$46,829,252 per additional ship. Dividing by 365, this difference is \$128,299 per day.

This means that if the sum of additional cargo inventory costs plus other additional operational costs of these ships (including the time charter) is less than \$128,299 a day, then case B is overall cheaper. One would initially think that such a threshold would be enough. But it turns out that this is not necessarily the case if in-transit inventory costs are factored in.

To compute in-transit inventory costs for this case, we hypothetically assume that cargo carried by these vessels consists of high value, industrial products (e.g., computers, luxury cars, or similar), whose average value at the destination (CIF price) is \$20,000/tonne. We also assume the cost of capital being 8%. This means that one day of delay of one tonne of cargo would entail an inventory cost of $I_c = PR/365 = 20,000 * 0.08 / 365 = \4.38 . This may not seem like a significant figure, but it is.

Computing the in-transit inventory costs for this case gives a total annual difference of \$200,000,000 (\$4,200,000,000-\$4,000,000,000) in favor of case A, which moves cargo faster. This figure is significant, of the same order of magnitude as the fuel cost differential.

Assuming also a time charter rate of \$25,000 per day (typical charter rate for a Panamax containership in 2007), the total other operational costs of the reduced speed scenario are \$958,125,000 per year for 105 ships, versus \$912,500,000 for 100 ships going full speed. Tallying up we find a net differential of \$11,478,741 per year in favor of Case 1, meaning that in-transit inventory and other operational costs offset the positive difference in fuel costs.

Of course, other scenarios may yield different results, and the reduced speed scenario may still prevail in terms of overall cost, under different circumstances. For instance, if the average value of the cargo is \$10,000/tonne, and everything else is the same, then the difference in annual inventory costs drops to \$100,000,000, rendering the reduced speed scenario a profitable proposition (with a total cost reduction of \$88,521,259 per year). Actually, speed reduction remains profitable if the value of the cargo is no more than about \$18,800/tonne (which can be considered as a break-even CIF price).

Many other variants of this scenario can be studied. For instance, we can investigate the effect of reduced fuel prices and reduced time charter rates (as was the situation in late 2008). If the price of fuel drops to \$250/tonne and the daily charter rate drops to \$15,000 a day, this speed reduction scheme is unprofitable for both the \$20,000/tonne and the \$10,000/tonne CIF cargo prices, perhaps reflecting the reduced potential for savings in fuel costs if fuel prices are low. In this scenario, the break-even CIF price of the cargo (below which reducing speed is profitable) will drop to about \$7,000/tonne.

How much would it cost to avert one tonne of CO₂ for the whole fleet for the previous speed reduction? We shall use Eq. 5.8, see Section 5.6.2.

For the containership example of Psaraftis and Kontovas (2009b), the CATC values for some of the scenarios examined are as follows (Table 5-3):

Table 5-3: Values of CATC (Psaraftis and Kontovas (2009b))

Scenario	CATC (\$/tonne of CO ₂ averted)
p=\$600/tonne P=\$20,000/tonne O _c =\$25,000/day	9.28
p=\$600/tonne P=\$10,000/tonne O _c =\$25,000/day	-71.56
p=\$250/tonne P=\$20,000/tonne O _c =\$15,000/day	104.94
p=\$250/tonne P=\$10,000/tonne O _c =\$15,000/day	24.10

This table confirms that CATC can vary widely. It is also interesting to note that the difference in CATC between the 1st and 2nd scenario is the same as that between the 3rd and 4th scenario (\$80.84/tonne in both cases). This is not a coincidence, and can be explained by the structure of expression 5.8. In these examples, the influence of in-transit inventory costs in the value of CATC can also be seen clearly. This means that perhaps one of the biggest obstacles that needs to be overcome is the unwillingness of the cargo owners to incur inventory costs for their cargoes. Optimized routing, logistics, and other operational measures that would reduce this inventory costs would be important.

As regards what threshold conceivably exists for CATC, that is, under what (positive) value of CATC a speed reduction scheme would still be considered desirable, this issue is currently open and it is not an easy one to address. For example, in a conceivable CO₂ Emissions Trading Scheme (ETS) for shipping, a monetary value would be put on a per tonne basis, for instance, \$30/tonne of CO₂ averted, and emissions reduction measures would be evaluated against such a threshold. Other possible thresholds were presented in Section 5.3. Note again, that besides constant market prices, damages from pollution are a non linear function and, thus, cost effectiveness ratios should be used with caution.

The concept of CATC, as defined above, can be generalized to measures other than speed reduction (see the Marginal Abatement Cost presented in this Section), and can be a useful concept for the evaluation of policy or other alternatives.

6

VALUATION FUNCTIONS INTO RISK ASSESSMENT: THEORY AND EXAMPLES

Values based on the previous Sections can be used as a cut-off rate for per unit costs to avert in the assessment of economic mitigation potentials; that is the cost-effectiveness threshold, the lamda (λ) value that was presented in Chapter 2. However, as discussed above the preferred way to access risks that have multiple effects is through Cost Benefit Analysis. Luckily enough, in maritime risk assessment it is very straightforward to convert CEA into CBA. Thus, values that were previously used as CEA thresholds can be used as pre unit prices when performing a CBA.

This Chapter presents a holistic approach to the risk assessment framework focusing on some useful insights on how to incorporate the valuation of the effects that were discussed in the previous three sections within risk assessment frameworks. Various examples are presented and, finally, the generalized theoretical framework is presented.

6.1 A simplified example

To start with, the following problem illustrates a simple way of using the non linear function in estimating the effectiveness of RCOs and is a modification of a problem presented in Kristiansen (2005).

Problem (Based on Kristiansen (2005)): A shore-based refinery receives crude oil from a small tanker. Risk analysis has shown that the probability of a spill during offloading is $\lambda=3 \cdot 10^{-3}$ spills per year and assume that there is only one such operation per year. The average size of the oil spill that may occur as a result of such an incident is estimated to be 100 tonnes. The cost of the spill (which is the benefit of averting an oil spill) is given by the following formula :

$$\text{Cost} = 51,432V^{0.728}(\text{in USD, if } V \text{ is in tonnes})$$

The refinery investigates the cost-effectiveness of an advance offloading installation that will reduce the probability of an accident to $1.5 \cdot 10^{-3}$ spills per year and its consequence to 50 tonnes (or 50% reduction in both probability and consequence). The cost of this device is 20,000 and has a lifetime of 10 years. Assume that currently there is no advance offloading installation and that the risk with and without the new installation is acceptable based on the ALARP principle. Examine if the implementation of the device should be recommended based on cost benefit analysis assuming a discount rate of 5% per year.

Solution:

The current risk is

$$R_o = P_o \times C_o = 3 \cdot 10^{-3} \left[\frac{\text{spills}}{\text{year}} \right] \times 51,432 \cdot 100^{0.728} \left[\frac{\text{USD}}{\text{spill}} \right] = 4,409.15 \left[\frac{\text{USD}}{\text{year}} \right]$$

After the new offloading installation the risk will be

$$R = P \times C = 1.5 \cdot 10^{-3} \left[\frac{\text{spills}}{\text{year}} \right] \times 51,432 \cdot 50^{0.728} \left[\frac{\text{USD}}{\text{spill}} \right] = 1,330.99 \left[\frac{\text{USD}}{\text{year}} \right]$$

which means a risk reduction of $\Delta R = R - R_o = 3,078.16 \left[\frac{\text{USD}}{\text{year}} \right]$

Then Present Value of the benefit of this new installation (that equals to the discounted damage cost of the oil spill if the oil spill occurred) is

$$\Delta R = PV(B) = \sum_{t=1}^{N=10} \frac{\Delta R}{(1+i)^t} = \Delta R \cdot \frac{(1+i)^N - 1}{i \cdot (1+i)^N} = 23,768.72 \text{ USD}$$

Obviously, this is more than the Net Present value of the implementation cost of the risk control option which is 20,000 USD and therefore the new offloading installation should be recommended.

6.2 Incorporating a non linear function within FSA

Currently all FSA studies assume a linear relationship between the cost of consequences and magnitude of consequences; thus there is an implied assumption of constant cost per unit of consequence. As discussed above the use of non linear functions to estimate the damage of oil spills has opened a new chapter for FSA. Some first thoughts on the way that a non-linear function can be used within FSA were expressed by Yamada and Kaneko (2010). However, things are easier said than done.

This Section will present some initial thoughts on the subject. Then an example will be given based on the FSA on tankers that was prepared by SAFEDOR and, finally, a generalization of this method will be proposed. Note that in this Section referring to 'FSA on tankers' means the FSA on crude oil carriers that was carried out by SAFEDOR and submitted to the IMO by Denmark, see doc. MEPC 58/17/2.

First of all, assume that the spill cost function is given by the formula produced after regression analysis of IOPCF data which is as follows (Kontovas et al., 2010):

$$\text{Cost (V)} = 51,432V^{0.728} \text{ (in USD, if V is in tonnes)}$$

The use of this particular function causes no loss of generality, as any other function of volume can be tried. These include the one used by Yamada (Japan), the one used by Psarros et al (DNV, Norway) or any other. Note however that the above function of ours was chosen by MEPC 60 as a test case for further analysis (see report of MEPC 60, agenda item 17). The function was chosen as it was judged as the most conservative among the 3 non-linear functions, that is, produces higher cost values among all 3 functions.

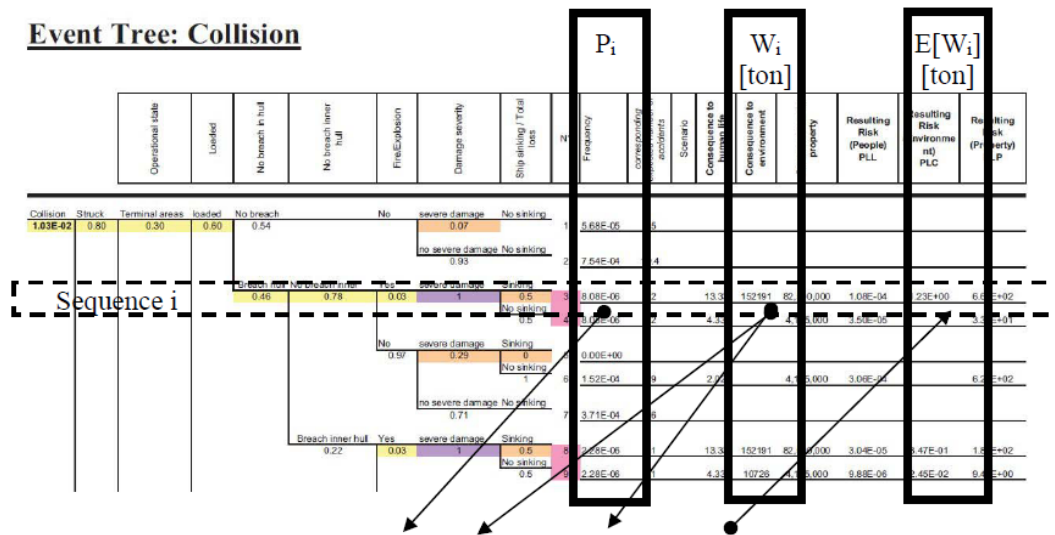
RCO evaluation by comparing the benefits (derived by using a function) and the costs is, in theory, presented in Psaraftis(2008) and Kontovas et al. (2010). Yamada (2009), Hammann and Loer (2010) and Yamada and Kaneko (2010) presented a way to incorporate a non-linear cost function within FSA. The latter paper forms the basis of a relevant submission to the IMO, see doc. MEPC 59/17/1 that was submitted by Japan. Both documents present a way that they could be used to calculate the benefit. However, in reality things are a little bit more complicated as it will be shown in the example that will follow, see Section 6.3.

In appendix 5 of the consolidated FSA Guidelines (see doc. MSC 83/INF.2) societal risk is defined as *"the average risk, in terms of fatalities, experienced by a whole group of people (e.g. crew, port employees, or society at large) exposed to an accident scenario"* and is usually expressed as **Potential Loss of Life (PLL)** which is defined as the expected value of the number of fatalities per year. In a similar way, the FSA on tankers defines **PLC (Potential Loss of Cargo)** as the expected quantity of released oil spillage in terms of tonnes per ship year.

The event tree structure of the FSA on tankers has a total of 82 sequential scenario branches with non-zero frequency, 20 scenario sequence branches of which are

associated with oil spill occurrence. For environmental damage, expected quantity of released oil spillage are also derived and summarized in the event tree model.

The following Figure is taken from the submission of Japan (doc. MEPC 59/17/1) and presents a way that the authors believe that a non-linear function could be used to estimate the benefits of the RCO.



$$Risk_i = E[C_i] = P_i \times (W_i \times CATS(W_i)) = E[W_i] \times CATS(W_i)$$

Note: Event tree is referred from document MEPC 58/INF.2, $E[C_i]$ denotes an expected costs[US\$] for sequence i. $E[C]$ [US\$] can be estimated by summing up $E[C_i]$ for all sequences

Fig. 6-1: Illustration of application of volume-dependent CATS in relation to an event tree
Source: doc. MEPC 59/17/1 and Yamada and Kaneko (2010)

Note that the original notations of doc. MEPC 59/17/1 are slightly different than those in our cost formulas. For example, the oil spill volume is denoted as W in the figure above and as V in our formulas.

According to Yamada(200x), risk $E[C]$ which is the expected benefit from averting an oil spill could be defined as :

$$E[C] = \sum (P_i \times C_i) = \sum (P_i \times W \times CATS(W_i)) = \sum (E[W_i] \times CATS(W_i)) \quad (9)$$

where P_i, C_i and W_i denote the probability of an oil spill, costs of the oil spill [US\$] and the oil spill size [tonnes] for every possible scenario i (=a sequence i of an event tree). CATS is the per unit total cost of the oil spill i , which when multiplied with the spill size gives the total cost of the oil spill.

First of all, note that this methods assumes that the λ threshold for assessing risk from oil spills (CATS) is assumed to be equal to the per tonne value of an oil spill. Furthermore, it is assumed that this value is not constant but depends on the oil spill size. More specifically it assumes a non-constant threshold. The CATS(W_i) is obviously the derivative of the cost function.

For each sequence the expected number of tonnes that will be averted is calculated as the product of the frequency of the event (P_i) and the average consequences (W_i) and is presented as $E[W]$. This is PLC value for each sequence. This value should then be multiplied with the per tonne cost (which is a function of the spill volume) to estimate the risk (denoted as $E[C_i]$) and by summing all the relevant sequences the total risk may be obtained. The expected cost should be estimated before the implementation of the RCO and after it.

According to Yamada and Kaneko (2010), an RCO can be regarded as cost-effective if the following formula is satisfied

$$\Delta B - \Delta S > 0$$

where ΔB is the benefit by implementing the RCO which is the risk reduction (in monetary units) and ΔS is the cost of implementing the RCO. ΔB is equal to the expected cost of an oil spill before the implementation of the RCO ($E[C_{org}]$) and after ($E[C_{new}]$).

Therefore, the criterion becomes $\Delta S < \Delta B = E[C_{org}] - E[C_{new}]$

In general, this is a correct approach although the procedure needs some further clarification. First of all, in reality, the discounted costs and benefits should be compared. Furthermore, in most FSA studies submitted to the IMO the event trees have not been recalculated. In most cases, the RCO was assumed to erase the risk (which means $E[C_{new}]$ equal to zero) or a risk reduction as a percent of the initial risk was estimated by expert judgment.

Finally, notice that the Yamada and Kaneko (2010) comment on the equivalency of the cost effectiveness criterion and the cost benefit criterion although it lacks theoretical evidence as the one presented in this work, see Chapter 2. In addition, the cost function (Cost(V)) could be used to estimate the expected benefit by multiplying the probability P_i with Cost (W_i). Using the non linear cost function or the per tonne cost function and multiplying with the expected oil spill yields the same result. However, it will be commented later that all the problems that the use of non-linear function carries can be eliminated by using a piecewise constant function. That way constant values will be derived for each volume range.

6.3 A hypothetical example based on the FSA on tankers

MEPC 58/INF.2 was submitted by Denmark and presents a high-level Formal Safety Assessment pertaining to large oil tanker ships that was carried out by the EU-funded project SAFEDOR. The study performs an identification and prioritization of the most important scenarios of large tanker accidents and continues with a detailed investigation of the causes and consequences of these scenarios. In addition, the risk level of the operation of crude oil carriers was assessed in order to identify high risk areas and to propose new risk control options (RCOs). It was found that the current risk levels of large oil tankers of double hull construction with respect to the potential loss of crew and oil cargo appeared to be within the ALARP region. Furthermore, a number of RCOs namely 7.1 (enhanced cargo tank subdivision), 7.2 (increased double bottom height) and 7.3 (increased side tank width) were found to be cost-effective for new-built tankers of all sizes considered (Panamax, Aframax, Suezmax and VLCC), with the exception of RCO 7.2 which is found not to be cost-effective for VLCCs. The reader is referred to the study for a full list of recommendations.

Our hypothetical example deals with contact, which is one of the scenarios evaluated in this FSA. Contact events consist of scenarios where the vessel accidentally comes into contact with a floating object or a fixed installation. Most contacts take place within congested waters with dense ship traffic, crossing routes and areas with large ship speed variations. The basic causes are because of bad visibility, navigational problems such as human errors or equipment failure such as radar failure, steering or propulsion failure. The following Qualitative risk model presents the event sequence of a contact of a tanker with a floating object (such as iceberg or a boy) or with a fixed installation (for example an offshore terminal or rocks).

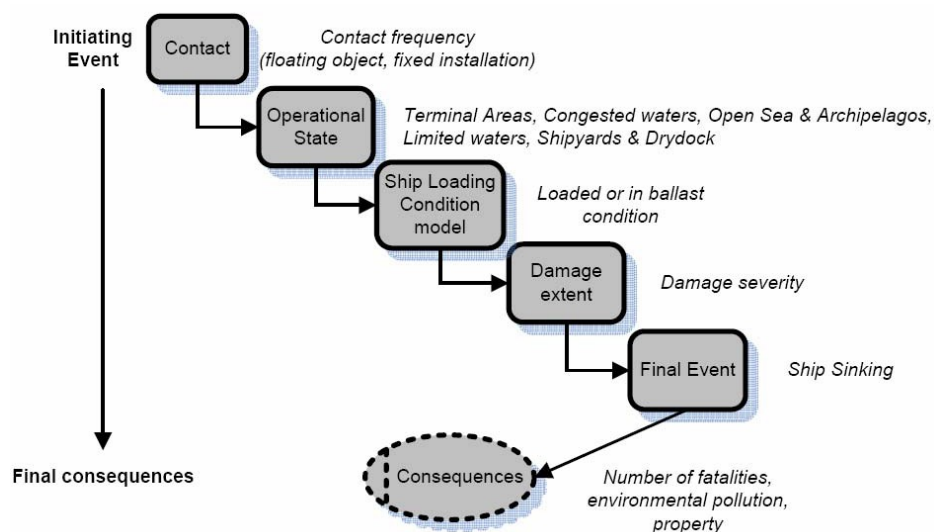


Fig. 6-2 : Event sequence in contact risk model of an Oil Tanker (Source: MEPC 58/INF.2)

The high-level event tree model for collision accidents has been elaborated on the basis of the qualitative and quantitative considerations and has event tree total of 52 sequential scenario branches with non zero frequency, 7 of which are associated with oil spill occurrence. For simplicity, the event tree with only these 7 branches is presented in the following Figure. Most probabilities are estimated by using expert judgment.

For the consequence part, the numerical average size of one tank (10,726 tonnes) is assumed to be the expected oil outflow in those scenarios, where "given the accident and the ship is assumed loaded, the inner hull is breached and there is a severe damage without ship sinking". For scenarios with non-severe damage the expected oil outflow was calculated as a percentage of DWT of the ships involved and was equal to 912.5 tonnes for contacts with a fixed installation and 0 for a contact with a floating object.

Risk analysis has found that the basic causes are mainly bad visibility, navigational and technical failures. One of the risk control options that could eliminate the risk of collision is a docking aid - a Terminal Proximity and Speed Sensor (appears as RCO 5 in the FSA on tankers). It is assumed by the analysts that there is no perceived reduction in the risk of fatality due to the implementation of a docking system. This RCO will lead to the elimination of the current risk –this means a risk reduction of 100%.

In terms of potential loss of cargo (PLC), calculations suggest that the RCO will prevent 1.4E-01 tonnes of oil per ship year or 4E+00 over the 25 year ship lifetime from being spilled, see also Figure 6-2.

Regarding the costs of the system, the FSA study on tankers says *“The cost of implementing a Doppler type docking system is largely associated with the initial purchase price which is considered to be \$70,000 based on industry figures provided by docking aid suppliers. Other perceived costs include an outlay of \$4,000 every five years for maintenance during dry docking periods, and an annual figure of \$400 for general spares and repairs. Over 25 years this provides a Net Present Value of \$85,840.”*

Based on the simple example above and the method that we described in the previous section, the cost effectiveness of this measure will be estimated. The benefit is assumed to be estimated by the non-linear formula proposed by Greece, see Kontovas et al. (2010).

For each sequence the risk is estimated as $R_i = P_i \times Cost(V_i)$ expressed in USD per year. The sum for all scenarios is the estimated benefit for averting the oil spills pertaining to contact. Thus, the penultimate column in Figure 6-3 presents the benefit of averting the average spill of each sequence based on the non linear formula and the last column based on a constant per tonne value (here assumed 40,000 USD per tonne).

CONTACT	Fixed Installation	Terminal areas	loaded	Breach hull	Breach inner hull	severe damage	No sinking	Ship sinking / Total Loss	N°	Frequency	corresponding expected number of accidents	Consequence to environment (tonnes)	Resulting Risk (Environment) PLC	BENEFIT based on non-linear function	BENEFIT (PLC * 40,000 USD per tonne)	
3.72E-03	0.68	Operational state	Loaded	No breach in hull	No breach inner hull	Damage severity	Ship sinking / Total Loss	N°	Frequency	corresponding expected number of accidents	Consequence to environment (tonnes)	Resulting Risk (Environment) PLC	BENEFIT based on non-linear function	BENEFIT (PLC * 40,000 USD per tonne)		
															0.56	0.6
		Limited waters	loaded	Breach hull	Breach inner hull	severe damage	No sinking	no severe damage	No sinking	6.29E-05	912.5	5.74E-02	462.19	2,295.27		
								0.38	0.6	0.5	0.22	0.7	1.90E-05	912.5	1.74E-02	139.89
		Congested water	loaded	Breach hull	Breach inner hull	severe damage	No sinking	no severe damage	No sinking	6.68E-06	10726	7.16E-02	295.15	2,865.19		
								0.02	0.6	1	0.22	1	0			
		Open Sea	STOP	0												
		Shipyards	STOP	0.04												
		Floating obj	Congested water	loaded	Breach hull	Breach inner hull	severe damage	No sinking	no severe damage	No sinking	9.90E-06	10726	1.06E-01	437.52	4,247.22	
									0.32	0.42	0.6	0.3	0.22	0.5	0	
		Limited waters	loaded	Breach hull	Breach inner hull	severe damage	No sinking	no severe damage	No sinking	1.96E-05	10726	2.10E-01	866.02	8,406.81		
								0.29	0.6	0.43	0.22	1	0			
TOTAL RISK											1.41	6,094.91	56,311.37			

Fig. 6-3: Estimated Potential Loss of Cargo - Data source : MEPC 59/INF.2

The Present Value PV(B) of the benefit for the 25-year lifetime can then be calculated using Eq 2-3.

Assuming a constant per tonne figure (40,000 \$/tonne) :

$$PV(B) = \sum_{t=1}^N \frac{\Delta R}{(1+i)^t} = 56,311.37 \cdot \frac{(1+5\%)^{25} - 1}{5\% \cdot (1+5\%)^{25}} = 793,649.29 \text{ USD}$$

By using the non linear formula proposed by Greece :

$$PV(B) = \sum_{t=1}^N \frac{\Delta R}{(1+i)^t} = 6,094.91 \cdot \frac{(1+5\%)^{25} - 1}{5\% \cdot (1+5\%)^{25}} = 85,901.31 \text{ USD}$$

The Present Value of the cost of implementing a docking system is associated with

1. the initial purchase price: \$70,000

$$PV(\text{initial}) = 70,000 \text{ USD } \$70,000$$

2. outlay of \$4,000 every five years for maintenance

$$PV(\text{outlay}) = \sum_{i=1}^5 \frac{4,000}{(1+5\%)^{5 \cdot i}} = 10,202.60 \text{ USD}$$

(payments on the 5th, 10th, 15th, 20th and 25th year)

3. an annual figure of \$400 for general spares and repairs

$$PV(\text{spares}) = \sum_{i=1}^{25} \frac{400}{(1+5\%)^{25}} = 400 \cdot \frac{(1+5\%)^{25} - 1}{5\% \cdot (1+5\%)^{25}} = 5,637.58 \text{ USD}$$

The total net present value of the cost is **PV(C) = 85,840.17 USD** (in line with the given value).

Cost Benefit Criterion

The net present value of the cost of the implementation of the RCO is \$85,840 and is less than the benefits based on the above calculations. Therefore the RCO should be considered for implementation. Note that the constant per tonne figure grossly overestimated the benefit of averting the oil spill.

Note that RCO5, in which our hypothetical example is based, has an effectiveness of less than 100%, which was assumed here. In this case, the correct way is to estimate monetized risk before and after the implementation of the RCO in order to estimate the benefit of implementing this RCO. It is obvious that if we performed such an analysis the result would be that the RCO5 is not cost effective since our hypothetical RCO that has 100% effectiveness **just** passes the cost benefit criterion.

6.4 Generalization: The analytical theoretic approach

We come back to the definition of risk. In maritime risk assessment 'risk' is normally presented *"as the product of the consequences and the probability of occurrence"* (Kristiansen,2005). Kaplan and Garrick (1981) define risk as a "set of triplets", a set of scenarios S_i , each of which has a probability P_i and a consequence X_i . Furthermore, their approach is to use the frequency with which an event might take place which is essentially the notion of uncertainty about the frequency which is the 'probability of frequency'.

Therefore, a risk analysis tries to answer the following questions (Kaplan and Garrick, 1981; Bedford, 2001):

- (i) What can happen,
- (ii) How likely is it to happen? and
- (iii) Given that it occurs, what are the consequences.

The first question is usually answered within the phase of the hazard identification. Note that hazards that are not identified will not be assessed therefore this first step is also a very crucial one, see also Kontovas and Psaraftis (2009).

STEP I. ESTIMATION OF PROBABILITY AND CONSEQUENCES

The second and third questions are the main focus of the risk assessment procedure (Step 3 in FSA process, see below). Hazards leads to consequences of different types, these may be related to human safety, to the environment and to the property, for example the ship itself. Therefore, it is important have an estimation of the probability and the consequences for each sequence of the event and for each effect. The methods that can be used include among other the construction of event and fault trees and Bayesian networks . The next step is to combine these two measures.

According to Vinnem (2007), estimating the fatality risks starts with calculating the so-called Potential loss of Life (PLL). One way to express PLL is by using accident statistics and define it *"as the number of fatalities experienced in a given period, usually per year"* (Vinnem, 2007). The notion of PLL is crucial in maritime risk assessment (Aven,2009; Kristiansen,2005). Kristiansen(2005) notes that PLL *"has the shortcoming of not incorporating any exposure measures"*. This is acknowledged in the FSA Guidelines and PLL is defined taking into account the exposed population; and PLL is most usually expressed in fatalities per ship year.

Furthermore according to Potential loss of life (PLL) per year is a term proposed by Shell and emphasizes that the fatalities are not inevitable with good safety management, but PLL tends to suggest (and is sometimes used for) the maximum number of fatalities in a single accident. PLL is also sometimes used for the expectation value of lifetime group risk (known here as the lifetime fatality rate). Therefore potential loss of life (PLL) is a widely accepted metric of risk and may represent the average number of fatalities expected in a given period or in some cases the maximum number of fatalities.

Hazards leads to consequences of different types, these may be related to human safety, to the environment and to the property, for example the ship itself. Assume that our analysis accesses M different types of consequences (j=1..M).

Fig. 6-4 illustrates the general case of an event tree with N possible scenarios where an initial event (that has a probability of λ^1) may lead to two possible events (E1 and E2). Each sequential event may happen ('yes') or may not ('no') so there are N main events. For each scenario (i=1..N), the probability of occurring of each consequence (λP_i) the same. However the consequences $C_{j,i}$ are different. Furthermore suppose that the benefit of averting such a consequence may be evaluated by the nonlinear function f_j which is a function of $C_{j,i}$.

Therefore, for each of the M effects (f.e human loss, oil spill, CO₂ emissions) the related risk may be calculated by adding the potential losses for each sequence i (i=1..N) as follows:

$$R_j = \sum_{i=1}^N \lambda P_i \cdot f_j(C_{i,j}) \quad \text{Eq. 6-1}$$

Summing all effects (and we can do this since we have a common monetary unit to estimate the damage) the total risk (usually expressed in USD per ship year) can be calculated as follows:

$$R = \sum_{i=1}^N \sum_{j=1}^M \lambda P_i \cdot f_j(C_{i,j}) \quad \text{Eq. 6-2}$$

¹NOT TO BE CONFUSED WITH THE CEA CUTOFF RATE (i.e THRESHOLD) λ

²The most fundamental distinction between risk' and 'uncertainty' in this branch of economic theory, is due to Knight (1921). A situation is said to involve risk if the randomness presents itself in the form of exogenously specified or scientifically calculable objective probabilities, as for example with gambles based on a roulette wheel or a pair of dice. A situation is said to involve uncertainty if the randomness is presents in the form of alternative possible events, for example as with bets on a horse race, or decisions involving whether or not to buy earthquake insurance (Wakker, 2005).

INITIAL EVENT	EVENTS		EFFECT 1				EFFECT 2	
	E1	E2	No	Probability	Consequences	Risk	Consequences	Risk
Initial Event	Yes	Yes	1	λP_1	$C_{1,1}$	$\lambda P_1 f_1(C_{1,1})$	$C_{2,1}$	$\lambda P_1 f_2(C_{2,1})$
λ	$P(E1)$	No	2	λP_2	$C_{1,2}$	$\lambda P_2 f_1(C_{1,2})$	$C_{2,2}$	$\lambda P_2 f_2(C_{2,2})$
		1- $P(E2 E1)$						
	$1-P(E1)$	Yes	3	λP_3	$C_{1,3}$	$\lambda P_3 f_1(C_{1,3})$	$C_{2,3}$	$\lambda P_3 f_2(C_{2,3})$
		No	4	λP_4	$C_{1,4}$	$\lambda P_4 f_1(C_{1,4})$	$C_{2,4}$	$\lambda P_4 f_2(C_{2,4})$
		1- $P(E2 not E1)$						
			\vdots	\vdots	\vdots		\vdots	\vdots
			N	λP_N	$C_{1,N}$	$\lambda P_N f_1(C_{1,N})$	$C_{2,N}$	$\lambda P_N f_2(C_{2,N})$

Figure 6-4 : A generalized event tree

STEP II. ESTIMATING RISK REDUCTION

In order to estimate the benefit of the RCO, the remaining risk after the implementation of the control option should be estimated. In many FSA studies, it is assumed that an RCO totally eliminated the risk (risk reduction of 100 percent). However, in most cases the risk reduction is estimated by expert judgment as a percentage of the original risk. It should be stressed out that the best way is to recalculate the event tree. Saying that an RCO will reduce the risk of an incident source let's say collision by, for example 10% besides being arbitrary, it is also very misleading. In any case, the benefit of the RCO is equal to the initial risk (before the implementation of the RCO) minus the remaining risk (risk after the implementation). The risk is expressed in currency per period – that is usually monetary units per year.

STEP III. ESTIMATING DISCOUNTED COSTS AND BENEFITS

The benefits of the reduced risk (ΔR) can be easily calculated for the T-year lifetime (T equal future amounts ΔR are incurred at regular intervals) of the Risk Control Option :

$$PV(B) = \sum_{t=1}^T \frac{\Delta R}{(1+r)^t} = \sum_{t=1}^T \frac{\sum_{i=1}^N \sum_{j=1}^M \lambda P_i \cdot f_j(C_{i,j})}{(1+r)^t} \quad \text{Eq. 6-3}$$

or, equally,

$$PV(B) = \frac{\sum_{i=1}^N \sum_{j=1}^M \lambda P_i \cdot f_j(C_{i,j})}{(1+r)^t} \cdot \left[\frac{(1+r)^t - 1}{r \cdot (1+r)^t} \right] \quad \text{Eq. 6-4}$$

where

- ΔR = Total risk reduction (currency)
- r = Discount rate per period (usually per year)
given as a decimal fraction (e.g. 5%=0.05)
- T = Number of periods (usually years)

The present value of the cost of the RCO may include initial purchase and installations cost (that are not discounted as they occur in year $t=0$) and other costs that may be annual ones and other periodic ones. According to Kristiansen(2005) - and this list is not exhaustive - these costs may include costs related to inspection and maintenance, operating costs, enforcement and administrative costs etc.

Usually, in order to estimate the present value of the benefits the equation of the present amount of a chain of equal future amounts (ΔR) is used. On the other hand, to estimate the present value of the costs, besides the installation and purchase costs that are not discounted the equation that calculates the present value of a future amount is calculated. For annual costs the equation that gives the PV of a chain of equal future amounts is used. Note again, that these basic equations are presented in Section 2.

STEP IV. APPLYING THE COST BENEFIT CRITERION

The net present value for each alternative, NPV, equals the difference between the present value of benefits and the present value of costs:

$$\text{NPV} = \text{PV} (B) - \text{PV} (C) \qquad \text{Eq. 6-5}$$

Rule 1: The basic rule for an alternative (relative to the 'status quo') is simple : *An option (risk control option) that has a positive NPV should be considered to be recommended for implementation.*

Rule 2: *Among alternatives to the current state ('status quo') the options should be ranked based on their NPV from the highest to the lowest under the constrain that the NPV should also be positive.*

Note also, that the NPV of a policy *'is identical to the present value of the net social benefits. Thus selecting the project with the largest NPV is equivalent to selecting the project with the largest present value of net social benefits'* (Boardman et al., 2007).

Furthermore, there are other similar criteria, for example the positive NPV criterion is equivalent to the criterion that the ratio of present value of benefits to the present value of costs should be greater than one. As discussed in Section 2, the NPV criterion is the one to be used. In case of budget constrain methods that are used in cases of capital rationing should be used.

Finally, as it has been emphasized it is important to conduct a sensitivity analysis to convey how sensitive the estimated net benefits are to changes in assumptions. Note that Kontovas (2005) and Kontovas and Psaraftis (2009) have commented on the great uncertainty that FSA faces especially in estimating the benefits. Costs are usually most easy to estimate and have in general market prices. On the other hand, valuing damages to human safety and the environment is rather complex. Note that techniques, such as the Monte Carlo sensitivity analysis may strengthen a risk assessment, see next Chapter.

7

CONCLUSIONS AND FURTHER RESEARCH

7.1 Discussion and Conclusions

Inevitably all human activities entail some risk. We have seen that the economic theory as regards efficiency suggests that the optimal risk is in general non-zero since moving towards zero risk entails huge economic costs.

This work has described the generic way to assess risk in goods that have mainly non-use value (such as to human health and the environment) and for which in many cases we can construct a hypothetical market but can be easily include those that have a market value (e.g. property). The focus has been on estimating the damage done to human health and the environment (mainly because of oil spills and ship air emissions). We assume also that this damage is equal to the benefit when implementing measures to reduce the relevant risk. Therefore, our work helps estimating the damage when an activity poses an extra risk and the benefit of acts that mitigate risk. In most cases, the effects are dealt separately but using our generic framework one can combine this different benefits by just adding the monetary benefits or damages. This is essential since an action to reduce a risk may increase another risk. The underlying technique (CBA) is nothing new, thus, the importance of our work is the introduction of non linear valuation function into practical risk assessment, the holistic approach (generalization), the proposal on the correct use (including the identification of the current pitfalls) of the relevant techniques and the proof that CBA is more appropriate technique. Another key point is that the framework is presented along with its strong foundations on the underlying economic theories.

Note again that much of the state-of-the work on the effects of oil pollution that has performed by the author, among others, has been submitted by Greece to the International Maritime Organization and has been incorporated into the proposal for the amendment of the Formal Safety Assessment Guidelines, see Appendix B.

The author had also the opportunity to assist his thesis supervisor (who had a key role in the relevant activity within MEPC) and participated in MEPC 62 which finalized the relevant work and incorporated some of the issues presented above into draft amendment of the FSA guidelines. The work within MEPC was very important since we gathered experience on how the issues described in the previous Chapters are addressed in practice. Thus, this experience was invaluable in identifying much of the pitfalls and the need for a more holistic and more solid approach as presented in this dissertation.

As a closing remark we hope that this work could help producing more solid results. But we stress out that the bottom-line results are just figures. Contrary to the current way of thinking within MEPC, risk assessment should be an input to risk-informed decision-making (Apostolakis, 2004). In the case of high uncertainty it may be dangerous to depend on a risk-based approach. Apostolakis (2004), who has an experience for over 40 years in nuclear risk assessment, comments that no Quantitative Risk Assessment (QRA) analyst cares about the bottom line numbers and that *"QRA is performed to understand how the system can fail and to prioritize the failure modes, not to produce a set of numbers"*. In addition he states that *"the decision that need to be made will be better if quantitative information that has been peer reviewed is available"*. Therefore, we note that maritime risk assessment is very important as it can provide useful engineering insights and may be used to prioritize areas that need to be focused and not to pay attention on risks that do not pose a real harm to society. The results of CBA should be carefully analyzed, a sensitivity analysis should be performed and the results should be peer reviewed if they should support decision-making. Finally, contrary to what is currently happening with Formal Safety Assessment, the bottom-line figures are to inform and support decision-making and not to directly impose the implementation of the measures that have been found to be cost-beneficial.

We summarize the key points of our work as follows:

- Most human activities entail some risk
- Measures to mitigate one risk may increase another
- Optimal risk is not zero as zero risk usually comes at a huge cost
- Valuation methods to attach monetary values to effects have been addressed
- These 'prices' are just tools to account for the change of the relevant risk
- Most of the effects have non market values; not to be though as a 'real' value to exchange something.

- Damage from an activity is equal to the benefit from averting the same amount of harm
- Damages are in general non linear functions of the amount of harm. This is addressed for the first time with practical maritime risk assessment.
- Damages or benefits can be simply added to provide the total effect (based on the utilitarian view)

- Methods for the economic appraisal of private projects and projects that affect the society in whole (mainly regulations) such as the Cost Benefit Analysis and Cost Effectiveness Analysis have been presented along with their roots in economic theory. Pitfalls and possible deficiencies have also been addressed.
- Given the strong non linearities in the damage functions, it may be difficult (if not impossible) to use Cost Effectiveness Analysis for some effects. Given that CEA is the main tool within FSA, which is the major tool used in maritime risk assessment this is a real behavioral change.
- The sole use of CBA is advocated within maritime risk analysis are a more generic tool and most capable to address the deficiencies of CEA. However, the correct application of CEA is presented for the case which it may be appropriate to do so noting that CEA and CBA would come to the same results under specific conditions.

- We shortly present, in theory, the different ways that these 'prices' may be estimated and dedicate whole 3 chapters (each for each effect; human health, effect from oil spills and carbon dioxide emissions) to investigate possible ways to derive these figures.
- We, then, illustrate using examples the ways that these valuation and evaluation techniques may be used in practice and most importantly within maritime risk assessment techniques (using FSA as such a typical method).
- We spread some seed of generalization that so that this framework can be used to assess multiple effects.

- Finally, we present some issues of further research on issues that may have been already addressed elsewhere by have seldomly incorporated in the practical maritime risk assessment and especially the FSA method.

Coming to an end, we feel that this research as summarized above addressed the problem identified in the Introduction and may, thus, be useful to produce more solid and economically sound risk assessments and not only to benefit firms but also the society. In addition, all underlying economic theory has been presented to strengthen the framework presented. It is also hoped that in the near future risk analysts will move towards a more holistic, CBA-based approach and the deficiencies of other techniques will be well understood by those that are supportive of alternative techniques. To that extent, the easiness of incorporating the framework proposed in our work into the practical risk assessment has been clearly presented.

However, there is always room for improvement. The rest of this Chapter presents some elements from sensitivity analysis that can be used to strengthen the whole analysis based on well established techniques that are, however, not often applied within maritime risk assessment, which we may still say is not very well developed yet. In addition, another Section presents some ways to visualize the key factors and results that may help to better communicate the results between the experts themselves and the experts, policy makers and the society. Given that the society in most of the cases has a different perception of risk (from the analysts) we present some elements of a behavioral risk assessment. Although the theoretical background exists it may be relatively difficult to use it within practical risk assessment; however we feel that something like this is important and will be a key issue in the future.

7.2 Sensitivity Analysis and Simulation

Recall the simple problem based on Kristiansen (2005) that was presented in Section 6.1 involving a shore-based refinery that receives crude oil from a small tanker. Risk analysis has shown that the probability of a spill during offloading is $\lambda=3 \cdot 10^{-3}$ spills per year and assume that there is only one such operation per year. The average size of the oil spill that may occur as a result of such an incident is estimated to be 100 tonnes. The cost of a potential oil spill (which is the benefit of averting an oil spill) is given by the following formula :

$$\text{Cost} = 51,432V^{0.728} \text{ (in USD, if } V \text{ is in tonnes)}$$

The cost-effectiveness of an advance offloading installation that will reduce the probability of an accident to $1.5 \cdot 10^{-3}$ spills per year and its consequence to 50 tonnes (or 50% reduction in both probability and consequence) was investigated. The cost of this installation was assumed to be 20,000 and has a lifetime of 10 years. The present value of the benefit of this new installation (that equals to the discounted damage cost of the oil spill if the spill occurred) was estimated to approximately 23,768 USD. The net present value of the investment is therefore 3,768 USD and therefore the new offloading installation was found to be a good investment.

The key factors in our analysis are two:

1. The assumption that both the probability and the consequence (in terms of the average expected size of the oil spill) will both be reduced by ΔR equal 50%
2. The cost of the installation will be 20,000.

Now, what happens if reduction in probability and consequence is not 50% ? After all, this is an expert judgment that can be false. And what happens, if the final cost of the installation is more than what we originally anticipated?

Sensitivity analysis can show how the net present value behaves in changes of the two abovementioned factors and can be used to study the uncertainty in the analyst's assumptions and conclusions in order to ensure the quality of the risk assessment. This kind of analysis can be applied in various ways within CBA (Boardman et al., 2001).

Simple sensitivity analysis is performed by estimating the NPV value for different possible values for the uncertain factors (ΔR and cost in our case). Worst and best-case analysis can also be used by considering the pessimistic and optimistic values of the factors in question. However, in most cases, more sophisticated techniques such as the well known Monte Carlo simulation can be used, see Boardman et al. (2001) and Vose(2009) for using Monte Carlo simulation in risk analysis and CBA.

In short, a Monte Carlo simulation is a technique that builds a model of possible outcomes by substituting a range of values (taken from a probability distribution) for any uncertain factor. During the simulation, values are randomly sampled from the input probability distributions and for each set of values (called an iteration) the output is calculated. This is performed for thousand times and the result is a probability distribution of possible outcomes, in this case the NPV.

The trial version of the @Risk software was used to perform a simulation for the example presented above. For the cost, a triangular distribution was used, with an most likely value of 20,000 USD, a minimum of 17,000 USD and a maximum of 25,000 USD. For the reduction in probability and oil spill size, a PERT distribution was used. This is similar to the triangular distribution since the analyst defines the minimum, most likely and maximum value, but values around the most likely are more likely to occur. This distribution is mostly used when a value comes from expert opinion, see Vose(2009). The PERT distribution that was used had a most likely value of 50%, a minimum of 45% and maximum of 55%.

The probability distribution of the NPV after 1,000 iterations is presented in Fig. 8-1a and the cumulative distribution in Fig. 8-1b.

The minimum NPV in the simulation was close to -2,067 USD and the maximum close to &,600 USD. The mean was 3,092.5 USD. In addition, in almost 95% of the cases the net present value was above 0, this means that we are confident that the investment is cost beneficial.

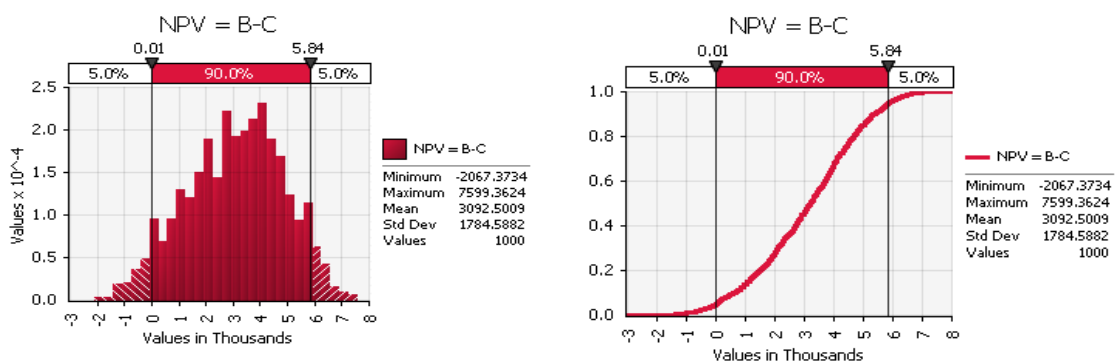


Fig.7-1: Monte Carlo Simulation – Results for the NPV

7.3 Visualization

Again, the following chapters discuss the way to economic appraisal of projects. Risk reduction measures or options, medical interventions, investments; these are all projects. Regardless of the techniques employed for ranking (prioritizing) and selection (decision criteria), the analyst has to display and communicate these data to the decision makers. Recall that risk communication is a part of the Risk Management Process (ISO 31000:2009) as presented in Fig. 7-1.

Common display mechanisms include four-quarant grids, X-Y charts, bar charts, matrices and bubble charts (Levine, 2005).

7.3.1 Bar Diagrams

Probably the easiest way to present the results of a CEA or a CBA is by using Bar Diagrams. Given that in most cases a large number of alternatives exists it is preferable to use horizontal bars rather than vertical ones. Bar diagrams are easy to be constructed but are however of limited use since they can only represent up to three different variables (when length, width and color of the bar are used).

In the case of Cost Effectiveness Analysis, a horizontal bar can be used for each RCO. The length of the bar can represent the marginal cost of avoiding a unit of the adverse effect (it has to be either one effect or one non monetized one) and the width can represent the number of units of the effect averted. In order just to compare the RCOs the width is not important. Within FSA, the length can display the GCAF or NCAF index of the RCO and the width the number of lives averted for the case of risk to human life.

A similar graph can be constructed to show the cost effectiveness of measures to reduce air emissions. The case where these measures are not mutually exclusive is presented in Figure 7-2. Similar bars can be constructed for all other effects (risks to human life, risk of oil pollution etc). Note that especially in the case of policies to reduce the effect on Climate Change the visualization of mutually exclusive options in vertical format is know as 'MAC curves' and are of extreme importance. As presented in Section 5.3.2, a MAC curve is a graph that shows the cost-effectiveness index of all the available measures (each measure represented as a bar) that are ordered in a function of increasing abatement costs. The height of each bar represents the average marginal cost of avoiding a ton of CO₂-eq given that all measures on its left are already applied and the width represents the potential of that measure to reduce emissions, see Section 5.3.2 for more.

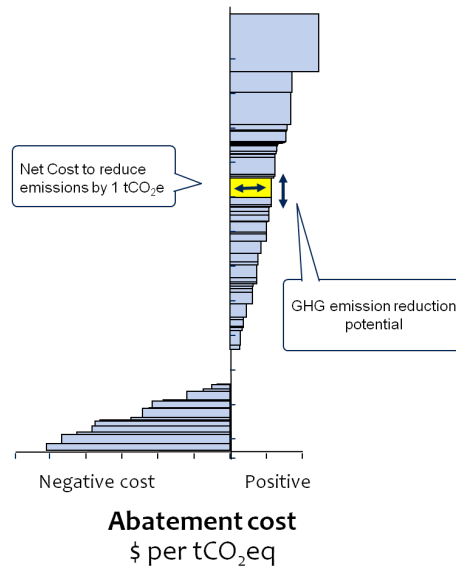


Fig.7-2: Bar Diagram to visualize risk control options

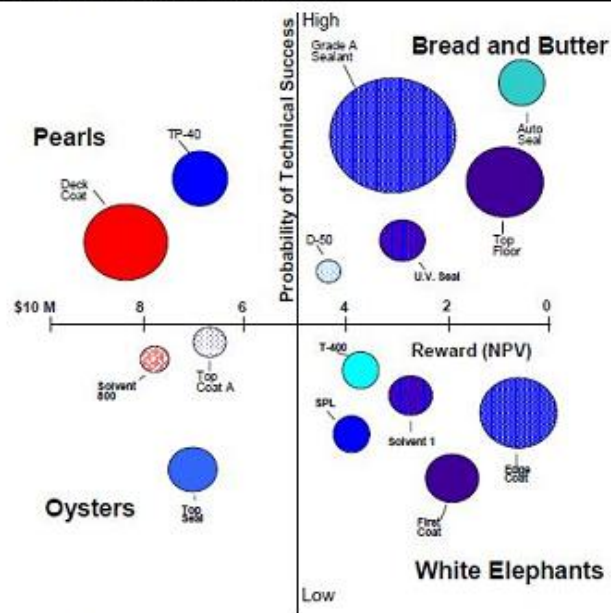
In the case of Cost Benefit Analysis the length of each bar (RCO) can be used to represent the net present value of the act. This is enough information to rank each alternative. In addition, the width of each bar can be used to represent the number of units averted by each project for a better overview of the alternatives.

7.3.2 Bubble Diagrams

Bubble diagrams (also called bubble charts) are of extreme importance, since they can effectively support multidimensional analysis of data, where each project can be displayed by a single bubble. Through the use of axes, variable bubble size, color, shape, shadow, line thickness, some bubble charts can display as many as seven different variables in a single chart. Visualization using bubble charts can be effective for a relatively small number of projects, mainly to avoid overlapping, and a small number of variables presented.

According to Cooper(2011), the most popular bubble diagram is the risk-return chart, being the method used by more than 40 percent of businesses that employ a systematic portfolio management method. Here, the x-axis measures the reward to the company (for example the net present value) and the y-axis the probability of success. Figure 7-3 comes from Cooper (2011) who presents a sample diagram for a chemical company. In this case the size of the bubble shows the annual resources committed to each project (that can be dollars per year or work-months allocated). According to Levine (2005) half of the spending of companies goes to the so-called 'Bread and Butter' -these are small projects with high likelihood of success but low reward (upper-right quadrant). Obviously, the most desirable projects are those in the upper-left quadrant (the so-called 'Pearl' projects).

The Popular Risk-Reward Bubble Diagram



Projects are plotted as bubbles on this two-dimensional risk and reward grid. The bubble sizes denote the resources committed to each project.

Source: Cooper, Edgett, and Kleinschmidt, "Portfolio Management for New Products"

Fig.7-3: The popular Risk-Reward Bubble Diagram -Source: Cooper (2011)

Recently, Abrahamsen and Aven (2011) presented some safety oriented bubble diagrams to be used in Project Risk Management. In the bubble diagrams presented in Fig. 7-4 the risks are considered through three dimensions: 1) consequence, 2) probability and 3) manageability (which they define as "potential for reducing risk and obtain desirable outcome"). An example of a risk included in this bubble diagram is 'ship collision next year'. The bubble illustrates the probability of such a event, the consequences associated (f.e the expected losses if the event happens) and the manageability of the event.

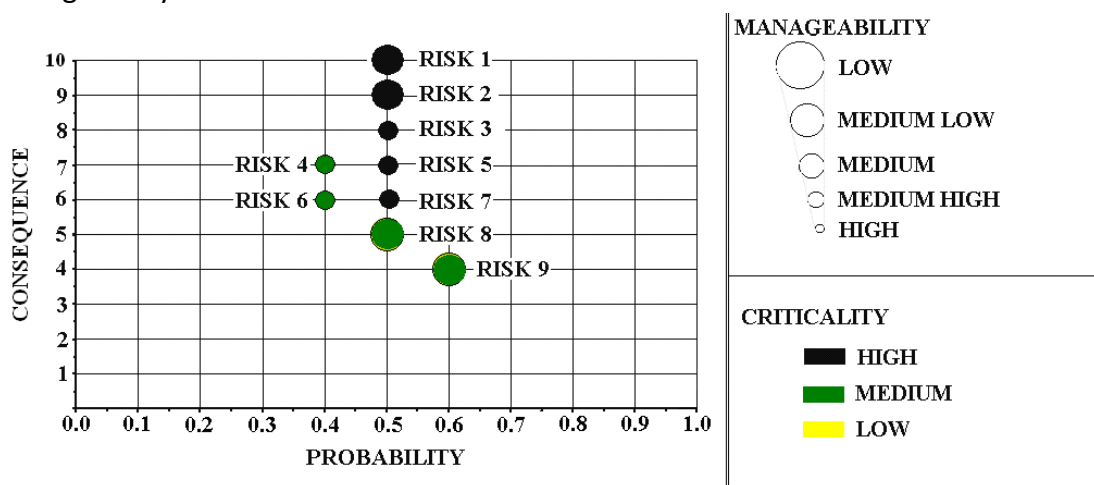


Fig.7-4: Sample safety oriented Bubble Diagram - Source: Abrahamsen and Aven (2011)

Bubble diagrams that present the information used in Cost-Effectiveness and Cost Benefit analysis and, therefore, more relevant to the work presented in this dissertation can also be constructed.

In the case of cost effectiveness analysis, the x-axis can present the number of units averted (f.e. fatalities, tonnes of oil or emissions), the y-axis the costs and the cost effectiveness ratio can be represented by the size of the bubble. In the case of net cost effectiveness index, the y-axis can present the net cost (that is the cost minus the benefit) and a different colour can be used for measures (bubbles) with negative ratios.

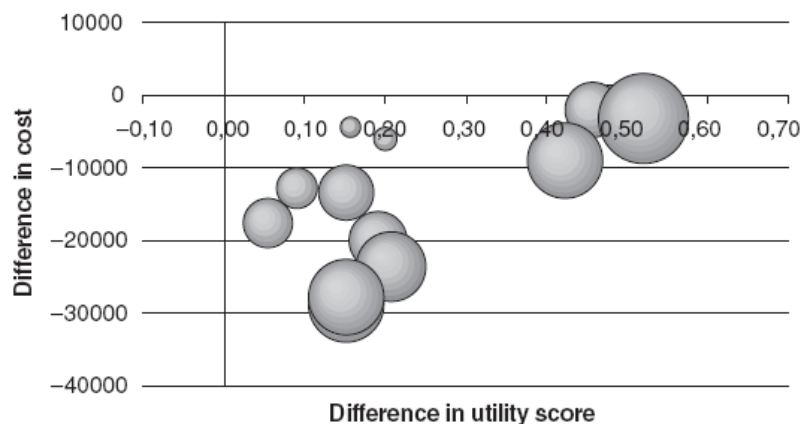


Fig.7-5: Risk Management Process ISO 31000:2009 -Source: ISO (2009a)

Recall that in health economics the cost effectiveness ratios are being extensively used and usually the differences in risk, cost and benefit between the action under consideration and the status quo (or similar actions) are used. The following figure (Fig. 7-5) presents the cost-utility of peritoneal dialysis (Sanchez et al.,2010). The bubble chart is plotted as follows: the horizontal axis presents the difference in mean utility scores, the difference in mean costs of their intervention programme compared to the conservative therapy are displayed along the vertical axis and the Cost-utility ratio is represented by the size of the bubbles.

First of all, a bubble diagram that illustrates the current hazards can be used in Hazard Identification (Step 1 in Formal Safety Assessment) to identify the areas where attention is needed. In this case, each bubble presents the current risk; that is the probability and consequence associated with each hazard. Bubbles in the upper left corner are associated with high risk and should be the main areas of investigation to identify possible risk reduction measures. In that case the information presented will be the same in the case of a risk matrix. As an extension, the bubble size may display the potential risk reduction that can be achieved if an RCO is implemented, or the maximum reduction achieved by all possible RCOs.

Table 7-1: Possible variables presented in bubble charts

#	DESCRIPTION	X-AXIS	Y-AXIS	SIZE	COLOR
1	Generic Cost Effectiveness Index	Risk reduction (in units averted) f.e fatalities, tonnes of oil, tonnes of CO2	Discounted Cost of RCO or Discounted Net Cost (i.e. Cost-Benefit) of RCO	Gross/Net Cost effectiveness ratio	Different for RCOs with negative ratio (mandatory)
2	Net Cost Effectiveness Index	Discounted Cost of RCO	Discounted Benefit of RCO	Risk reduction (in units averted) f.e fatalities, tonnes of oil, tonnes of CO2	
3	Cost Benefit Analysis	Discounted Cost of RCO	Discounted Benefit of RCO	Net Present Value	Different for RCOs with positive NPV
4	Cost Benefit Analysis	Discounted Cost of RCO	Discounted Benefit of RCO	Risk reduction (in units averted)	Different for RCOs with positive NPV
5	Cost Benefit Analysis	Risk reduction (in units averted) Effect A	Risk reduction (in units averted) Effect B	Net Present Value	Different for RCOs with positive NPV
6	Cost Benefit Analysis	Risk reduction expressed in non monetary units (f.e fatalities)	Risk reduction expressed in monetary units	Net Present Value	Different for RCOs with positive NPV

In the context of this work, various bubble charts can be illustrated, see Figure 7-6 for some hypothetical examples and Table 7-1 for the various possible different variables that can be presented by each bubble. These diagrams can be presented in a quantitative or qualitative form. In some cases, dashed lines can be used to display the acceptable regions (see for example the case of NCAF and GCAF criteria or values of the B/C ratio greater than 1) and different colors for accepted values (see for example, positive NPV values that satisfy the CBA criterion)

It is believed that the above can not only help to illustrate the various results but also assist in the communication of the results among the risk analysts but also them and the decision makers.

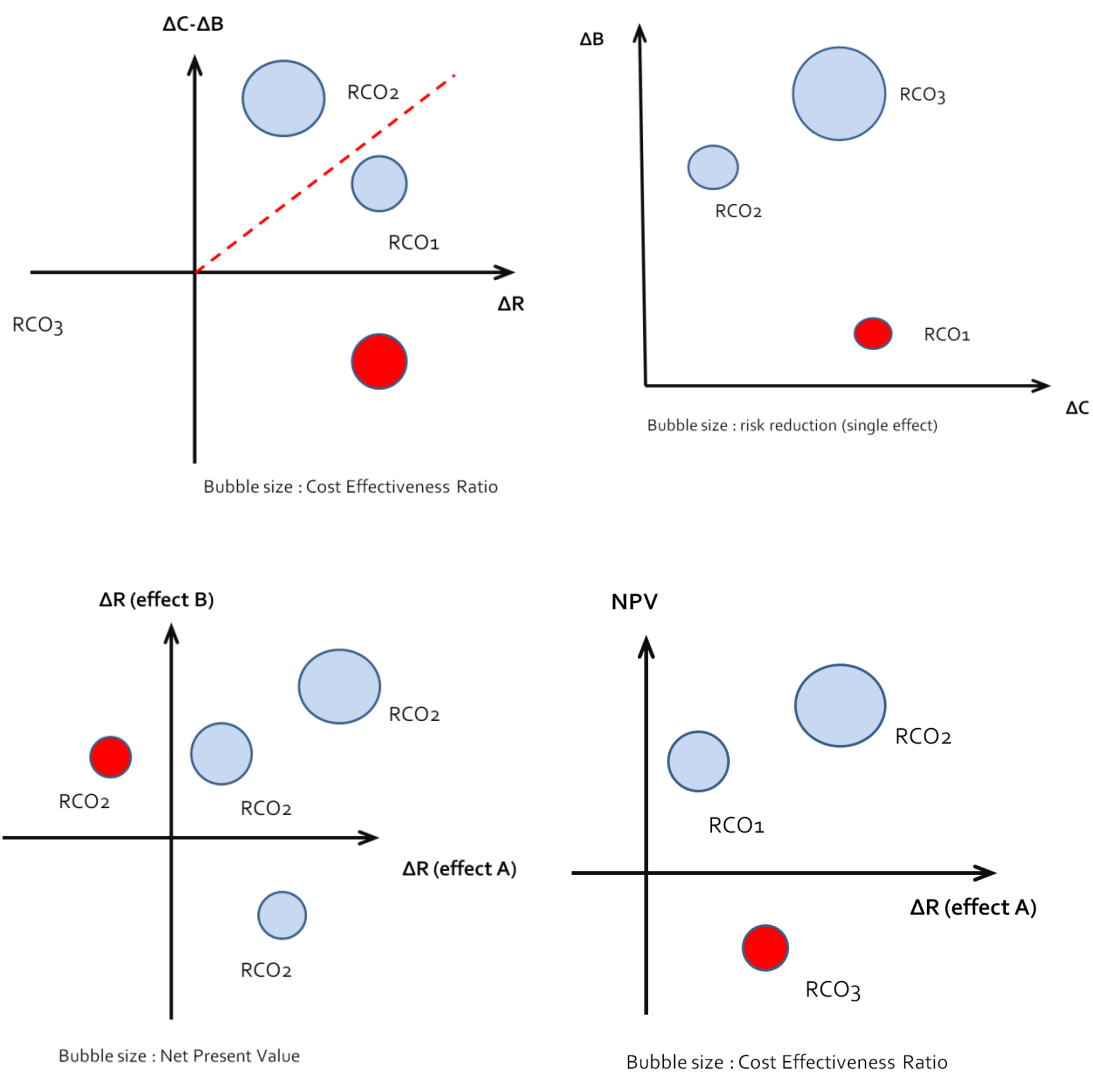


Fig.7-6: Some hypothetical CEA/CBA related Bubble Diagrams

7.4 Behavioral Risk Assessment

Cox (2008) presents an example to support his thesis that risk rankings do not necessarily support good decisions of resource allocation. The example identifies 3 different risk reduction opportunities:

- A. Act A that reduces risk from 100 to 80. It costs \$30.
- B. Act B that reduces risk from 50 to 10. It costs \$40.
- C. Act C that reduces risk from 25 to 0. It costs \$20.

This example can be constructed so that “all acts starts from the same base level of risk, say 50, and A,B and C reduce risk by 20,40 and25, respectively” (Cox, 2008).

According to the FSA Guidelines, the cost benefit assessment (Step 4) of Formal Safety Assessment may consist of estimating and comparing “the cost effectiveness of each option, in terms of the cost per unit risk reduction by dividing the net cost by the risk reduction achieved as a result of implementing the option” (Stage 4) and “rank the RCOs from a cost-benefit perspective in order to facilitate the decision-making recommendations”. Following this rationale, the cost per unit of risk reduction is calculated for each act.

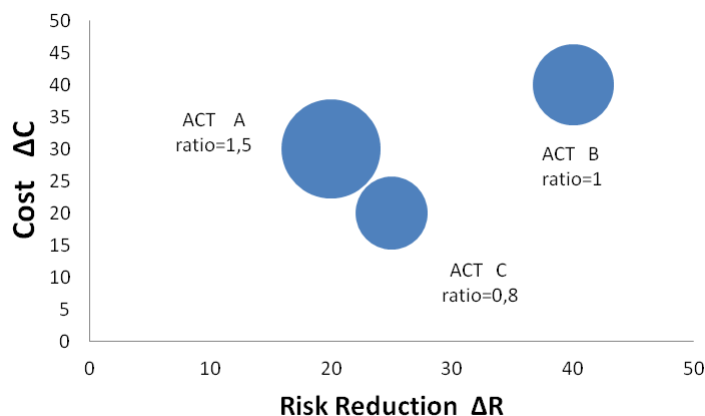


Fig.7-7: Bubble Diagram to be used for risk ranking

In the Bubble Diagram above the bubble size illustrates the costs per unit of reduction. One could translate the current FSA criteria as follows: ‘the smaller size the better’. Based on that, C is preferred than B and A than C. However, Cox (2008) correctly argues that given a budget constrain, this ranking does not lead to the optimal allocation of resources using the following rationale. For a budget of \$40, the largest risk reduction is achieved by B. At \$50 funding A and C achieves the greatest reduction. Finally, for a budget of \$60 best option is to fund B and C. Both CEA and CBA can be used to choose the optimal solutions. For the decision rules applied in CEA, see Section 2.2.5. However, it is much easier to choose the optimum mix of actions by using integer programming.

The optimal resource allocation for a set M of mutually exclusive projects can be replaced by the following linear programming problem (LP)

$$\begin{aligned} \text{maximize: } \sum x_i \quad \text{subject to: } x_i &= \begin{cases} 1, & \text{if project is implemented} \\ 0, & \text{if not implemented} \end{cases} \\ & \sum x_i C_i \leq B \end{aligned}$$

where e and c are the effectiveness (e.g. lives averted, Quality Adjusted years, tones of CO2 or oil averted or any units of effect) and the cost of project i . Note that this rationale can be extended if a percent of each project is implemented but this is out of the scope of this work.

Now, imagine that the cost for Act 3 is \$50 instead of \$20. Priority ranking based on the cost per unit even under no budget constrains ranks Act B last. However, how many decision makers would prefer C more than A? I assume many. Personally if I had to choose among these three projects I would definitely choose C. That is not an unusual behavior.

The following example, due to Zeckhauser, illustrates a similar behavior. Suppose you are compelled to play Russian roulette, but are given the opportunity to purchase the removal of one bullet from the loaded gun. *“Would you pay as much to reduce the number of bullets from four to three as you would to reduce the number of bullets from one to zero?”* Most rational people would be willing to pay much more for a reduction of the probability of death from 1/6 to zero than for a reduction from 4/6 to 3/6.

Research on the behavior of people has supported the *“preference for reducing a small risk to zero over a greater reduction in a larger risk”*, a heuristic bias known as the ‘zero-bias’ effect. In addition, the psychological ‘certainty effect’ implies that a reduction from 0.1 to 0 may be more valued than a reduction from, say, 0.4 to 0.3 since it *“replaces uncertainty with certainty”* (Cox, 2005). However, note that zero risk may not exist in practice for many activities due to technological and economical constrains. It has been also empirically verified that people are willing to pay disproportionately more to decrease higher mortality risk.

Therefore, we could argue that decision makers and lay people probably value more reductions towards zero. This is not the only bias in decision making. Research has proved that lay people and decision makers are "heuristically biased" in many ways when deciding which safety measures have to be prioritized (Slovic et al., 2000). In those cases welfare economics are clearly obliged to stray from the risk-neutral position and the expecting utility maximization.

7.4.1 Brief Introduction to Risk Perception

Behavioral economics and non-expected utility theories (such as PT) in particular have presented various heuristics and biases that may as well apply in risk analysis. There is no doubt that these should be taken into account when performing a societal CBA and a risk assessment which tries to identify policy measures. Regulations are to benefit the society and it is the society's perceived values that should be taken into account. In addition, these biases may help recognize some of the common problems identified when performing risk analysis, f.e when expert judgment is involved. Indeed, Slovic (1987) states that *“those who promote and regulate health and safety need to understand how people think about and respond to risk. Without such understanding, well-intended policies may be ineffective.”*

We shall now briefly focus on risk perception, which is the subjective judgment that lay people make about the characteristics of a risk. There are mainly three types of theory have been developed to explain the differences in the estimates of risk between people: psychology approaches (heuristics and cognitive), anthropology/sociology approaches (cultural theory) and interdisciplinary approaches (social amplification of risk framework).

We note that there is a huge amount of literature on this issue, which is out of the scope of this work to present. A key early paper was written in 1969 by Starr used a revealed preference approach to identify risks that may be considered acceptable by society. In addition, early psychometric research by Kahneman and Tversky identified some heuristics that may lead to inaccurate judgments, see the following Table. Furthermore, social amplification is triggered by the occurrence of an adverse which sometimes extend far beyond the direct damages to victims and property and may result in massive indirect impacts such as litigation against a company, increased regulation of an industry, and so on (Kasperson et al., 1988).

Noting that it is out of the scope of this work to present the relevant theories we describe some major heuristics and biased that are of interest within our work.

Firstly, in Table 7-2 we present the three general purpose heuristics (availability, representativeness and anchoring and adjustment) that were originally identified by Kahneman and Tversky (1974) in the early psychometric research.

Table 7-2: Biases and Heuristics presented in Kahneman and Tversky (1974)

Biases or Heuristics	Description (based on early work of Kahneman and Tversky)
Representativeness	<p>A heuristic according to which people estimate the probability or frequency of a hypothesis by considering how much the hypothesis resembles available data.</p> <p>Probability theory dictates that the probability of the conjunction of two sets must be less than or equal to the probability of being either of the sets. However, research indicated that people judged the conjunction as being more probable. In addition, the representativeness heuristic is related to the tendency of people to ignore prior statistical information (base-rate frequencies) when making assessments about probabilities which is contrary to the Bayes' Theorem.</p>
Availability heuristic	<p>Events that can be more easily brought to mind or thought of are judged to be more likely than events that could not easily be imagined. Note that it is also easier for us to recall information which has recently arrived.</p>
Anchoring and adjustment	<p>Anchoring and adjustment involves <i>"starting from an initial value that is adjusted to yield the final answer. The initial value, or starting point, may be suggested by the formulation of the problem, or it may be the result of a partial computation. In either case, adjustments are typically insufficient"</i> (Slovic & Lichtenstein, 1971; Tversky & Kahneman, 1974)</p>

Secondly, some heuristics and biases that are related to Prospect Theory, which was described in the previous Section, are illustrated in Table 7-3.

The reader is referred to Slovic(2000; 2011), two excellent books that cover most essential knowledge on risk perception. In the domain of referred journal papers some interesting readings are as follows :Regarding risk perception see Slovic and Fischhoff (1982), Pigeon et al. (1992). Human reasoning and decision biases have been repeatedly demonstrated in empirical studies (see Kahnemman, Slovic, & Tversky, 1982; Slovic, Lichtenstein, & Fischhoff,1988; Tversky & Kahneman, 1986). Moreover, see Kasperson (2005), Pigeon et al. (2003) for the social amplifications of risk.

Finally, this area has also been addressed within the context of Formal Safety Assessment (and mainly as regards expert judgment) in Skjong and Wentworth (2001), Rosqvist and Tuominen (2003) and Giannakopoulos et al. (2007).

Table 7-3: Biases related to Prospect Theory : Based on (X)

Biases or Heuristics	Description
Endowment Effect	The tendency of decision-makers to place a higher value on objects they own than on objects they do not. It explains why people rarely exchange a product they have already purchased for a better product. (Kahneman, Knetsch, and Thaler 1990).
Loss Aversion	The tendency of decision-makers to prefer avoiding losses versus acquiring gains. In risk assessment it is associated with risk aversion and with the fact that WTP is less than WTA for the same good.
Pseudocertainty Effect	The inclination to make risk-averse choices if the expected outcome is positive, but make risk-seeking choices to avoid negative outcomes (Tversky and Kahneman 1981; Slovic, Fischhoff, and Lichtenstein 1982).
Zero-Risk Bias	The preference for reducing a small risk to zero over a greater reduction in a larger risk. Individuals may prefer small benefits that are certain to large ones that are uncertain.

7.4.2 Non-expected Utility Theory within CBA and FSA

Now, recall that under risk neutrality people calculated the expected monetary value (EMV) of each option and chose the one with the highest EMV. The well-known St. Petersburg Paradox is historically important because it led Bernoulli (1738) to develop expected utility.

For many years (probably since the early eighteenth century) the classic economic theory suggested that people follow the norms of expected utility maximizing as described by the Expected utility theory (EUT) that was first introduced by Nicholas Bernoulli (Bernoulli, 1738) in his famous resolution of the St. Petersburg Paradox and was formally axiomatized by von Neumann and Morgenstern (1994) in their development of the game theory. EUT states that when making decisions under risk people choose the option with the highest utility, which depends on the potential outcomes, x_i , the subjective utility of each outcome, $U(x_i)$, and the probability of occurrence of each outcome, p_i as described by the following equation:

$$EU(O) = \sum p_i u(x_i)$$

Over the years, EUT has been challenged by many scientists including Allais (1952), who, in his path-breaking analysis, showed that the assumption that utility is linear in probabilities is not always satisfied and that actually decision makers disproportionately put more weight to low-probability high-consequence events. In short, Allais's (1953) famous example challenged the principle of linearity by showing that the difference between probabilities of .99 and 1.00 has more impact on preferences than the difference between 0.10 and 0.11. This paradox directly falsifies independence (one of the axioms of EUT), and is the most well-known proof that EU can be violated.

In addition, a substantial body of psychological research, mainly on biases (some of which will be presented below), shows that decision makers systematically violate the basic tenets of normative theories such as the Expected Utility Theory.

It is out of the scope of this work to present all the alternatives to expected utility (for reviews, see Camerer, 1989; Fishburn, 1988; Machina, 1987). We shall present the most influential one: prospect theory (Kahneman and Tversky, 1979; Tversky and Kahneman, 1992).

7.4.3 Prospect Theory (PT)

Prospect theory (PT) suggests that people put subjective weights both on values and probabilities and that people associate the weight of these values and probabilities with positive outcomes (i.e, gains) differently from those associated with negative outcomes (i.e, losses). Whereas EUT defines the value of a choice on the outcomes (the final asset), PT defines them on the prospects (gains and losses).

According to Prospect Theory, the subjective value PTV of the prospect O (which depends on the potential outcomes, x_i) can be determined as follows:

$$PTV(O) = \sum \pi(p_i)u(x_i)$$

where $\pi(\bullet)$ is the weighting function of the objective probabilities and $u(\bullet)$ is the weighting function of the objective probabilities, different for gains and losses.

Note that the original presentation of PT deals with prospects with at most two nonzero outcomes; i.e., prospects of the form p_1x_1, p_2x_2, p_30 , although this can be extended to multiple outcomes.

The value function is defined on deviations from a reference point, which is in line with the work of Markowitz (1952) that presents outcomes as positive or negative deviations from a “neutral reference outcome”, which is assigned a value of zero.

However, unlike Markowitz the value function proposed follows the commonly S shape and is normally concave for gains (implying risk aversion), commonly convex for losses (risk seeking) and is generally steeper for losses than for gains (loss aversion) (see Figure 7-8a below). These properties have been supported in a numerous studies of choice under risk that involve monetary outcomes (Fishburn and Kochenberger, 1979; Kahneman and Tversky, 1979).

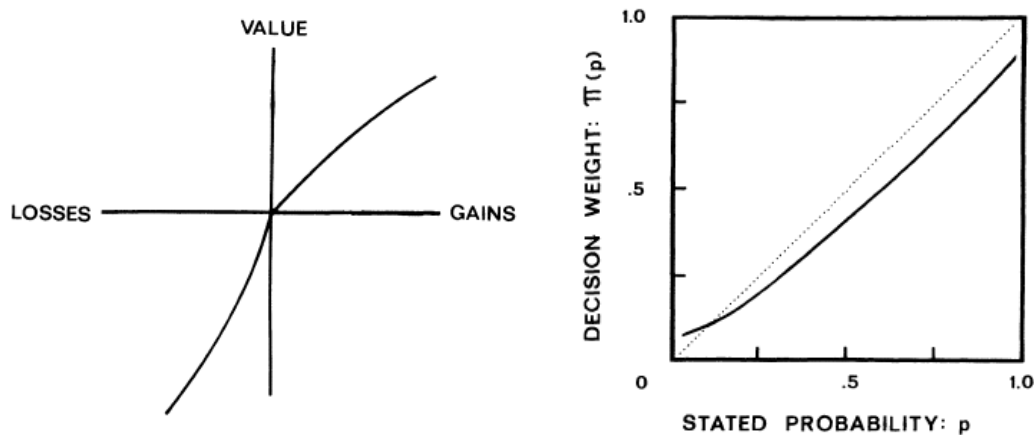


Fig.7-8: Hypothetical 'value function' and 'probability function' (Kahneman and Tversky, 1979)

In addition, note that the use of deviations rather than final outcomes is in line with the work of Markowitz (1952) that presents outcomes as positive or negative deviations from a "neutral reference outcome", which is assigned a value of zero. However, unlike Markowitz the value function proposed follows the commonly S shape (see Figure 7-8a), convex below the reference point, and concave above it.

In Prospect Theory probabilities are replaced by decision weights in order to obtain what we may refer to as 'subjective probabilities'. Decision weights are generally lower than the corresponding probabilities, except in the range of low probabilities (see Figure 8-2b for a typical shape of the weighting function). Decision weights are not probabilities: they do not obey the probability axioms (they sum does not add to 1) and they should not be interpreted as measures of degree or belief as they measure the impact of events on the desirability of prospects, and not merely the perceived likelihood of these events (Kahneman and Tversky, 2979).

Note that the original version of PT is concerned with simple prospects of the form $(x, p; y, q)$, which have at most two non-zero outcomes. However, the extension to prospects with any number of outcomes is straightforward. Kahneman and Tversky (1979) note that when the number of outcomes is large additional editing operations may be invoked to simplify evaluation. In addition, although the original PT involves

choices with monetary outcomes, the theory can be applied *“to choices involving other attributes, e.g., quality of life or the number of lives that could be lost or saved as a consequence of a policy decision”*.

7.4.4 Cumulative Prospect Theory (CPT)

Tversky and Kahneman (1992) superseded their original prospect theory with cumulative prospect theory (which they confirmed by a series of experiments) that employs cumulative rather than separable decision weights, applies to uncertain as well as with any number of outcomes and it allows different weighting functions for gains and for losses. According to Tversky and Kahneman, the original PT encounters two problems related to the weighting function: it does not always satisfy stochastic dominance (an assumption that many theorists are reluctant to give up) and readily extended to prospects with a large number of outcomes. These problems can be addressed by the rank-dependent or cumulative functional, first proposed by Quiggin (1982) for decision under risk and by Schmeidler (1989) for decision under uncertainty and form the basis of alternative non expected utility theories, the most prominent of which is the Rank-Dependent Utility Theory (RDUT). According to Wakker (2005), this extension of Prospect theory comprises the *“happy marriage”* between the empirical insights of Kahneman & Tversky (1979) and the theoretical insights of Gilboa (1987) and Schmeidler (1989) and *“at this moment of writing, 30 years after its invention, prospect theory is still the only theory that can deliver the full spectrum of what is required for decision under uncertainty, with a natural integration of risk (known probabilities) and ambiguity (unknown probabilities).”*²

According to Cumulative Prospect Theory, the subjective value PTV of the prospect O (which depends on the potential outcomes, x_i) can be determined as follows:

$$V_{CPT}(O) = \sum w(p_i)v(x_i)$$

where $w(\bullet)$ is the weighting function of the objective probabilities and $u(\bullet)$ is the weighting function of the objective probabilities.

²The most fundamental distinction between risk' and 'uncertainty' in this branch of economic theory, is due to Knight (1921). A situation is said to involve risk if the randomness presents itself in the form of exogenously specified or scientifically calculable objective probabilities, as for example with gambles based on a roulette wheel or a pair of dice. A situation is said to involve uncertainty if the randomness is presents in the form of alternative possible events, for example as with bets on a horse race, or decisions involving whether or not to buy earthquake insurance (Wakker, 2005).

The key elements of this theory are:

- 1) a value function that is concave for gains, convex for losses, and steeper for losses than for gains and
- 2) a nonlinear transformation of the probability scale, which overweights small probabilities and underweights moderate and high probabilities.

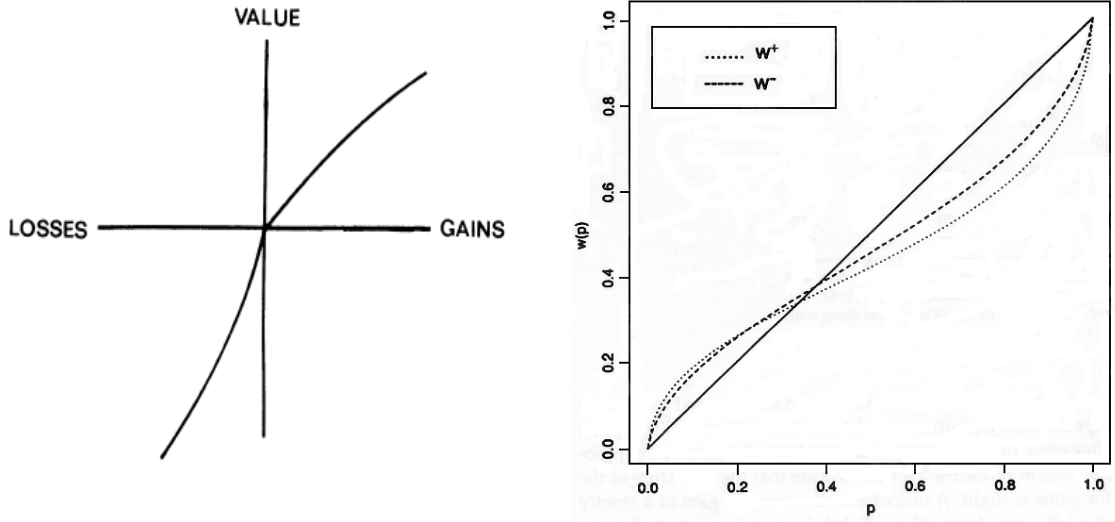


Fig.7-9: Weighting functions for gains (w^+) and losses (w^-) based on median estimates of γ and δ Source: Tversky and Kahneman (1992)

Where p is the probability and γ and δ constants, the probability weighting functions for gains and losses respectively are:

Losses :

$$w^+(p) = \frac{p^\gamma}{(p^\gamma + (1-p)^\gamma)^{\frac{1}{\gamma}}}$$

Gains:

$$w^-(p) = \frac{p^\delta}{(p^\delta + (1-p)^\delta)^{\frac{1}{\delta}}}$$

The value function (where the loss aversion parameter is λ) is defined as follows:

$$v(\alpha) = \begin{cases} \alpha^\theta & \text{if } \alpha > 0 \\ 0 & \text{if } \alpha = 0 \\ -\lambda(-\alpha)^\theta & \text{if } \alpha < 0 \end{cases}$$

Where $u(\alpha)=\alpha^\theta$ is used for gains and a different power function $u(\alpha)=-(-\alpha^\theta)$ for losses.

Tversky and Kahneman (1992) after empirical research resulted in the following median values: $\gamma=0.61$, $\delta=0.69$, $\theta=\theta'=0.88$ and $\lambda=2.25$.

A procedure for calculating the PT value of a prospect is as follows (Wakker, 2005):

- step 1. Completely rank outcomes from best to worst.
- step 2. Determine which outcomes are positive and which are negative.
(Steps 1 and 2 together determine the complete sign-ranking)
- step 3. For each positive outcome, calculate the gain-rank g .
- step 4. For all resulting gain-ranks, calculate their w^+ value.
- step 5. For each positive outcome α , calculate the marginal w^+ contribution of its outcome probability p to its rank; i.e., calculate $w^+(p+g)-w^+(g)$.

Treat losses symmetrically; i.e.,

- step 6. For each negative outcome, calculate the loss-rank ℓ .
- step 7. For all resulting loss-ranks, calculate their w^- value.
- step 8. For each negative outcome b , calculate the marginal w^- contribution of its probability q to its loss-rank; i.e., calculate $w^-(q+\ell)-w^-(\ell)$
- step 9. Determine the utility of each outcome (taking $U = \lambda u$ for losses).
- step 10. Multiply the utility of each outcome by its decision weight.
- step 11. Sum the results of Step 10.

The value of an outcome based on CPT is obviously not that easily calculated by hand. However, in the scope of this work we deal with cases where we choose between measures that reduce risk. So we either use the probability function for gains (WTA) or the one for losses (WTP) which significantly reduce the calculations needed.

7.4.5 Beyond the Expected Value

In brief, non expected value and utility (such as the theory presented above) should be used to calculate the perceived risk given a value function different for gains and losses and a nonlinear transformation of the probability scale noting that it is not easy to arrive at some commonly accepted functions. These perceived values (cost and benefits) should be the input to the Cost Benefit Analysis.

In addition such an approach could be used in risk frameworks to incorporate decision-maker's (and the society's) risk aversion or risk-seeking attitude in the utility (loss) function used. Note that in engineering context, risk is often linked to the expected loss. In mainstream engineering risk analysis mainly the expected values are taken into account but this is misleading for rare and extreme events (Haimes, 2004; Aven, 2011). In our case, a marine accident may have extreme consequences and relatively low probability. To that effect, it is, for example, pointless to use the regression lines derived for the case of oil spills to estimate the damage cost (or the cost to avert) a major oil spill such those caused by the 'Exxon Valdez' or the 'Prestige'. Therefore one should be very cautious when trying to assess the risk of extreme cases.

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A

CAUSES, TRENDS OF OIL SPILLS AND REGRESSION ANALYSIS OF IOPCF COST DATA

Appendix A1-1 Causes of Oil Spills

Since 1974 the International Tanker Owners Pollution Federation (ITOPF) has maintained a database of more than 10,000 oil spills from tankers, combined carriers and barges. The ITOPF database includes all reported accidental spillages (except those resulting from acts of war) which are divided into three categories by size (less than 7 tonnes, between 7 and 700 tonnes and greater than 700 tonnes). It should be noted that the figures for the amount of oil spilt in the following analysis present the total oil lost to the environment (including the oil one that is burnt and the one that remains in a sunken vessel). According to ITOPF (Huijer (2005) and ITOPF(2009)) the majority of reported spills are from small operational spillages of less than 7 tonnes for which complete reporting is difficult and, thus, 'little statistical reliance is placed on the data'. For this reason, our analysis we will focus on spills above 7 tonnes.

Table A1-1: Incidence of Spills by Cause, 1974-2008 - Source: ITOPF (2009)

CAUSE	<7	7-700	>700	>7	TOTAL
OPERATIONS					
Loading/Discharging	2825	334	30	364	3189
Bunkering	549	26	0	26	575
Other operations	1178	56	1	57	1235
ACCIDENTS					
Collision	175	303	99	402	577
Grounding	238	226	119	345	583
Hull Failure	576	90	43	133	709
Fire/Explosion	88	16	30	46	134
Other	2188	152	26	178	2366
TOTAL	7817	1203	348	1551	9368

Table A1-1 presents the number of incidences of spills categorized by cause for the period 1975-2008. Consequently, Figure A1-1 visualizes the data from this table by showing the percentages of each cause for the spill size categories and for spills over 7 tonnes.

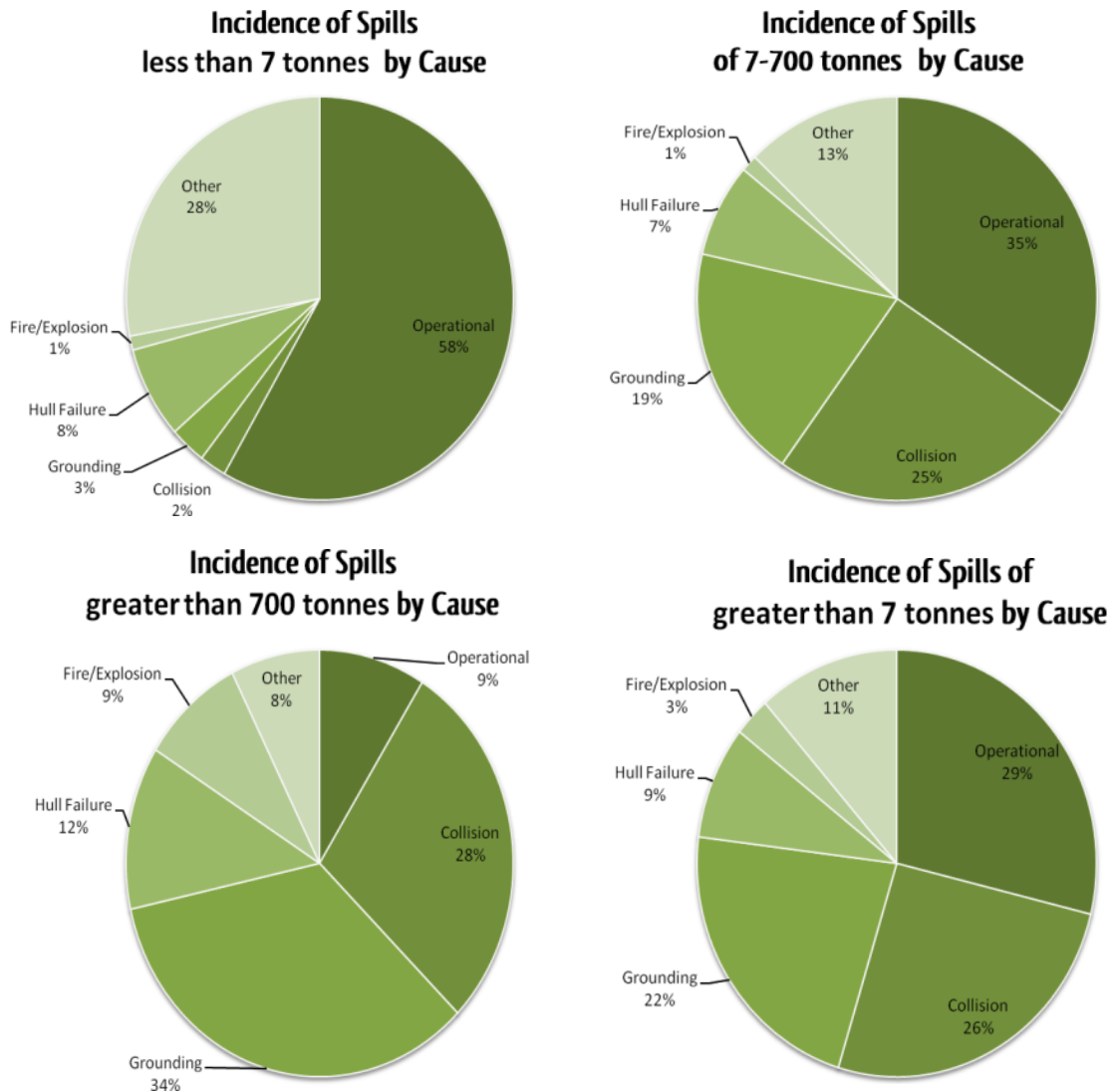


Fig. A1-1:Percentage of Incidence of Spills by Cause, 1974-2008 (based on data from ITOPF(2009))

It is apparent from the above figure that most spills from tankers result from routine operations, which normally occur in ports and oil terminals. The majority of these spills are small (less than 7 tonnes). Of the accidental causes, most incidents are from collisions and groundings. These results are in line with Huijer (2005) and Burgherr (2007). However, we should stress that the categories considered (Collision, Explosion/Fire, Grounding, Hull/Structural Failure and Other) are the same with those used by Lloyds’ LMIU, IMO, ITOPF and other casualty databases. Many such databases are more useful for aggregate statistical analysis of casualty data and less useful to draw conclusions as to the real cause of an accident and the sequence of events related to it. The latter may actually be a complex task to

ascertain, as it may be the object of an accident investigation that may take years to complete, not to mention that it may be the outcome of a litigation process that can be equally as long. Another drawback of databases such as the above is that root cause information is usually missing, as such information can only be retrieved after considerable analysis of the accidents themselves. Working with casualty databases that have incomplete or even wrong cause information may skew the ensuing analysis, particularly regarding measures to reduce risk.

The problem of lack of sufficient and relevant casualty data and especially the lack of data on the root causes of accidents in the existing databases has been repeatedly stressed (see Kontovas and Psaraftis (2009)). For example, databases categorize casualties as collision, grounding, fire, explosion, foundering etc. But these are consequences, not causes. Something always happened before a grounding or a collision occurred, and focusing on the consequence and not the root cause may skew the analysis that follows, and (at minimum) shift regulatory focus to measures that mitigate the consequence rather than prevent the cause.

Lack of real cause information in casualty databases remains a serious hindrance to researchers and risk analysts. Because of unclear or misleading cause information, the recommended Risk Control Options (RCOs) tend to be geared toward mitigating the consequences of accidents instead of preventing the accident in the first place (i.e. RCOs may be proposed so that the vessel better withstands a collision). However, if sufficient data was available for the root cause of the collision, better and possibly much more cost effective RCOs could be proposed (hypothetically duplication of a small mechanical part whose failure may lead to loss of power / steering). Most importantly, the accident could be prevented.

In this context, we draw attention to recent IMO doc. MSC 86/19/1 by Germany, and to doc. MSC 86/17/1 by Greece, that make this important point (among others), a point which has (to our knowledge at least), received far less attention than what it deserves. For a discussion of issues pertaining to uses of casualty databases, see also Devanney (2008).

Appendix A1-2 Trends in the Number and Volume of Oil Spills

As we discussed before, the majority of accidental spills on ITOPF’s dataset fall into the category of spills less than 7 tonnes. Furthermore, the number of spills per year for accidental spills of less than 7 tonnes are not presented by ITOPF. Figure A1-2 shows the number of spills per year and, as one may notice, the dominant size category is that of 7-700 tonnes. It is also apparent that there has been a decrease in the total number of spills as we will prove later on.

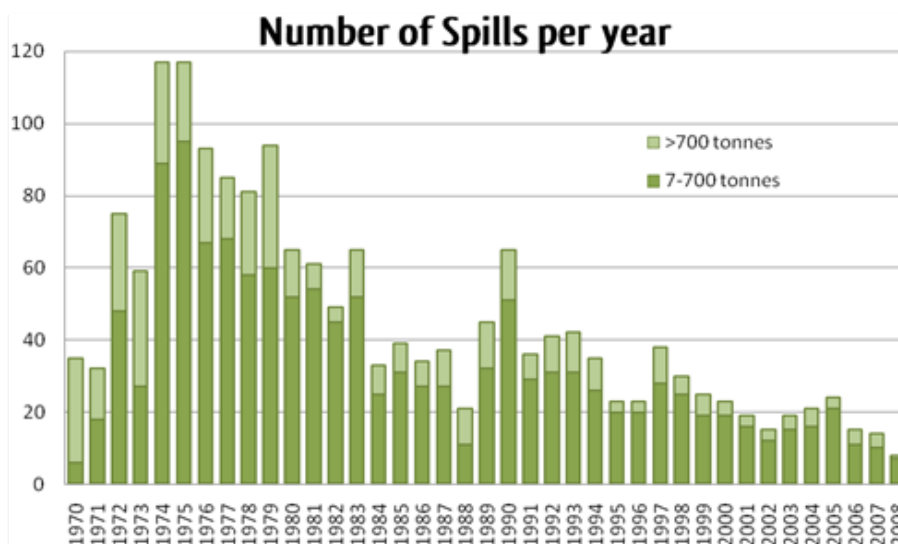


Figure A1-2: Annual Quantity of Oil spilt, 1974-2008 (based on data from ITOPF(2009))

The same downward tendency is apparent in the total annual quantity of oil spilt during the last decade (see Fig. A1-2). After the accidents of tankers Heaven (144,000 t) and ABT Summer (260,000t) in 1991, no accident above 100,000t has happened and, thus, the total amount of oil spilt decreased continuously. Figure A1-3 presents the average spill size per year and as one may notice after 1991 all average spill sizes are below 4,000 tonnes.

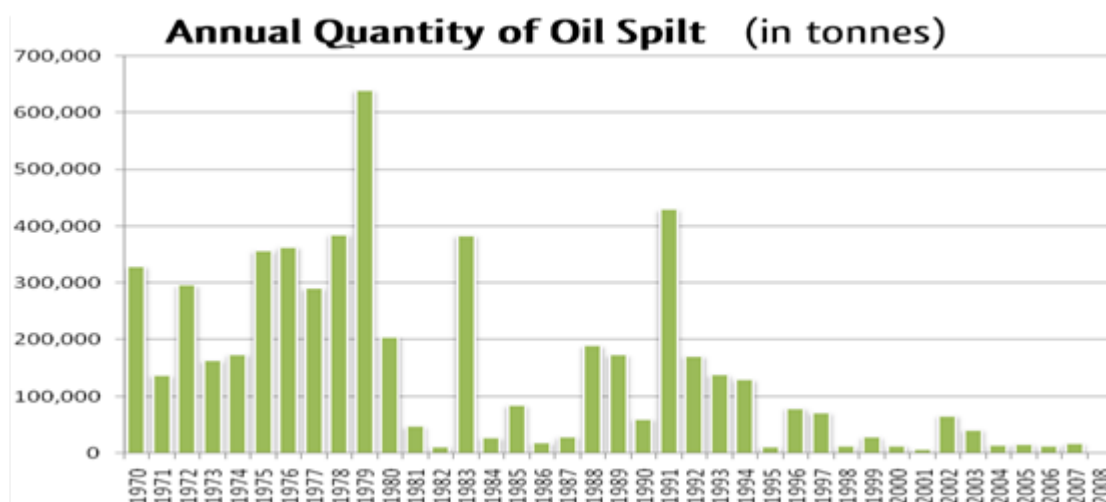


Figure A1-3: Annual Quantity of Oil spilt, 1974-2008 (based on data from ITOPF(2009))

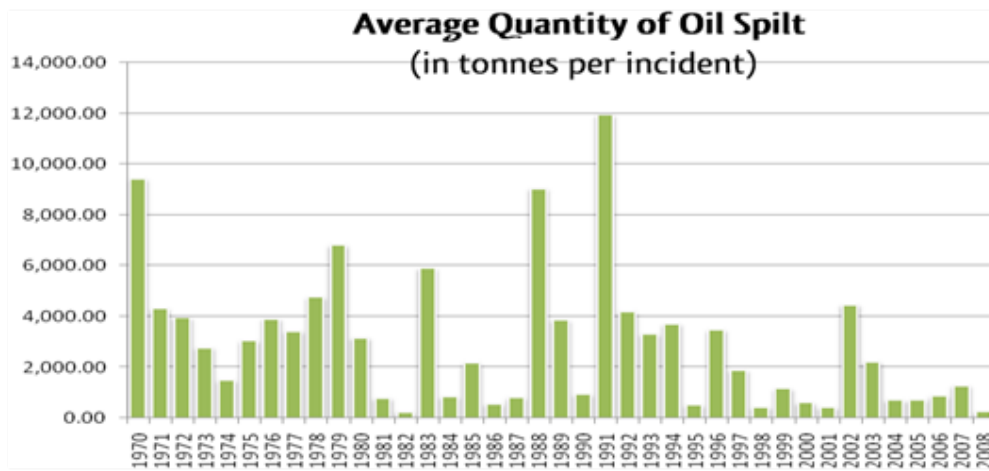


Figure A1-4: Average Quantity of Oil spilt per Incident, 1974-2008 (based on data from ITOPF(2009))

According to ITOPF(2009) “the incidence of large spills is relatively low and detailed statistical analysis is rarely possible,” thus, emphasis is only given on identifying trends (see Huijer (2005) and Burgherr (2007)). Burgherr (2007) provides an in-depth analysis of accidental oil spills from tankers if oil spills above 700t for the period of 1970-2004. Besides the trends identified in that paper, a global analysis addressing spatial autocorrelation of oil volumes across all locations using Marsden Squares (10° latitude by 10° longitude) was performed to detect if data are clustered or distributed randomly in space. Furthermore, the distribution of tanker spill volumes, and more precisely, the contribution of large spills to total volume was investigated using Lorenz curves. We support the previous comment by ITOPF and feel that the only statistical analysis that makes real sense is the identification of a possible trend.

The statistical analyses we performed to determine a possible trend included the calculation of the Autocorrelation Function (ACF) and Kendall’s τ for the three time series (see Fig. A1-5). Our analysis regards the last 30 years (period 1978-2008).

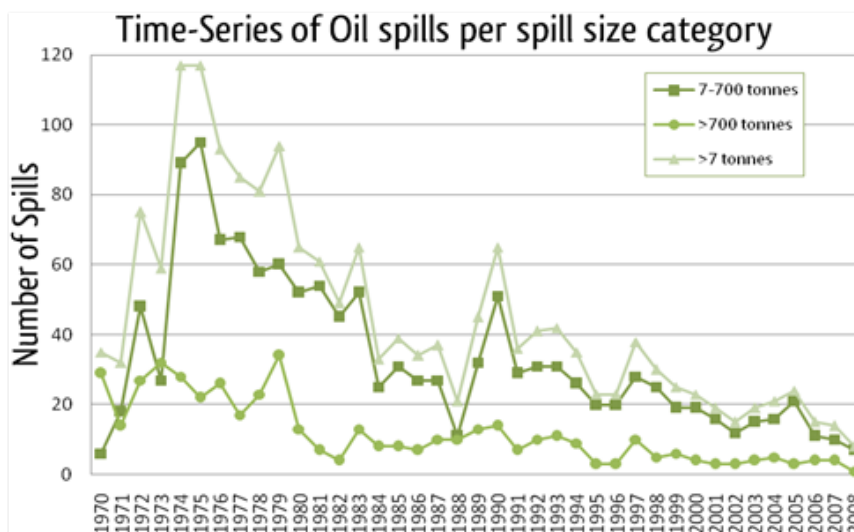


Figure A1-5: Number of Oil Spills per Year, 1970-2008 (based on data from ITOPF(2009))

The Autocorrelation Function

The Autocorrelation Function (ACF) at lag k for a given time series reflects the linear dependency between values that are separated by k time lags. The estimate for the ACF at lag k for an evenly spaced series, x_t , (as the ones we have here) of length n can be calculated by the following formula (see Jenkins and Watts, 1968 :

$$r_k = \frac{\sum_{t=1}^{n-k} (x_t - \bar{x})(x_{t+k} - \bar{x})}{\sum_{t=1}^n (x_t - \bar{x})^2}, k > 0$$

where \bar{x} is the estimated mean of the x_t series. The value or r_k can range from -1 to +1. The ACF constitutes a method for interpreting trends in data. Lag are the time periods separating the ordered data that are to calculate the autocorrelation coefficients. The maximum number of lags (as suggested by Box and Jenkins) is roughly N/4 for a series with less than 240 observations. Furthermore, T - test statistic used to determine if the autocorrelation for a particular lag equals zero or not. One commonly used rule is that a t-statistic greater in absolute value than 2 indicates that the corresponding autocorrelation is not equal to zero.

Based on this, we will investigate the presense of a possible trend in the number of spills for the size categories (7-700 tonnes, greater than 700 tonnes and their union, that is, greater than 7 tonnes) during the last 3 decades.

From 1978 to 2008 the number of observations are N=31, thus 8 is the number of lags that is used in our analysis. Figure A1-6 presents the ACF for the three size categories at the 5% significance level. The Autocorrelation Function (ACF) and T-test for each lag are presented in Table A1-2.

Table A1-2: Autocorrelation Function (ACF) and T-test values, 1978-2008

	7-700 t			> 700 t			> 7 t	
Lag	ACF	T		ACF	T		ACF	T
1	0.70098	3.90		0.54437	3.03		0.68494	3.81
2	0.52961	2.09		0.14812	0.65		0.47941	1.92
3	0.43082	1.50		0.05919	0.26		0.38688	1.39
4	0.34783	1.13		0.18313	0.80		0.34542	1.17
5	0.17294	0.54		0.15601	0.66		0.18844	0.61
6	0.07681	0.24		0.05870	0.25		0.10736	0.34
7	0.15946	0.49		0.04812	0.20		0.17174	0.55
8	0.09913	0.30		0.02230	0.09		0.09003	0.29

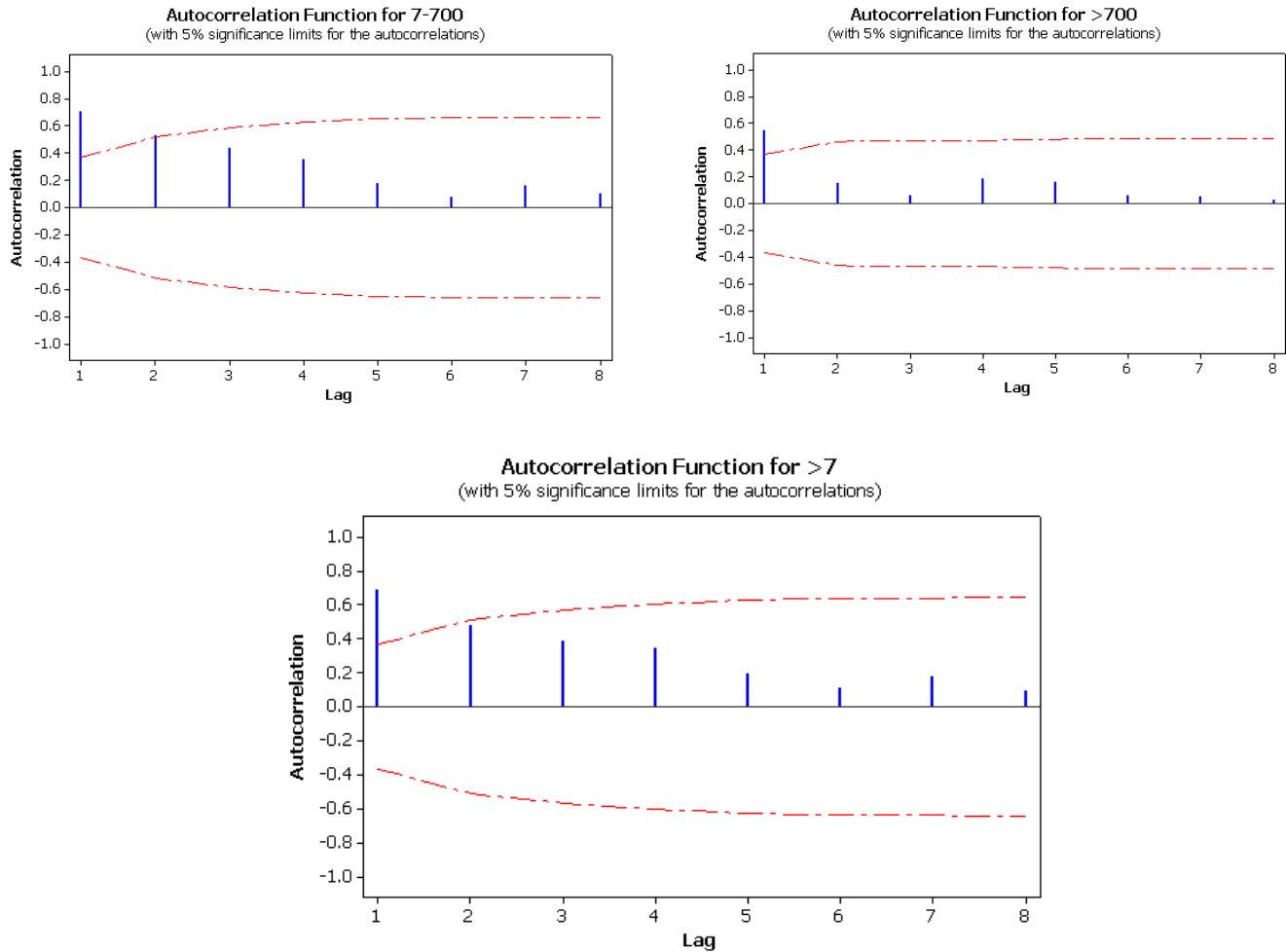


Figure A1-6: Autocorrelation Function (ACF)

The Mann-Kendall test

In environmental impact analysis scientists believe that the statistical characteristics of messy data make it difficult to use distribution-dependent test and, thus, nonparametric tests are mainly used. A nonparametric test is a method of testing a hypothesis, for example the null hypothesis, H_0 , the we will use according to which the sample data set does not possess a specific property which in our case is a trend. The nonparametric test that will be used is the Mann-Kendall test that was presented in Mann (1945) and is a test for randomness against time. This test is a special application of Kendall’s correlation test (Kendall, 1975) and is also known as Mann-Kendall or Kendall’s τ test.

Letting $x_i, i=1$ to n , be a sequence of measurements over time, the test examines the null hypothesis H_0 that the data come from a population where the random values are independent and identically distributed. The alternative hypothesis, H_1 , is that the data follow a monotonic trend over time. The Mann-Kendall statistic S (under H_0) is defined as follows :

$$S = \sum_{i=1}^{n-1} \sum_{j=i+1}^n \text{sgn}(X_j - X_i)$$

Where X_j are the sequential values, N the length of the data set and

$$\text{sgn}(x) = \begin{cases} +1, & x > 0 \\ 0, & x = 0 \\ -1, & x < 0 \end{cases}$$

The statistic S is approximately normally distributed with the mean and the variance as follows:

$$E[S] = 0$$

$$\text{Var}(S) = \frac{1}{18} \left[N(N-1)(2N+5) - \sum_{i=1}^N t_i i(i-1)(2i+5) \right]$$

where t_i is the number of ties of extent i .

The Mann-Kendall test (z) and the corresponding p -value are given by

$$Z = \begin{cases} \frac{S-1}{\sqrt{\text{Var}(S)}}, & S > 0 \\ 0, & S = 0 \\ \frac{S+1}{\sqrt{\text{Var}(S)}}, & S < 0 \end{cases}$$

$$p = 0.5 - \Phi(|Z|) = 0.5 - \frac{1}{\sqrt{2\pi}} \int_0^{|Z|} e^{-\frac{t^2}{2}} dt$$

If the p -value is small enough, the trend is quite unlikely to be caused by random sampling. In that case, a positive Z value indicates an upward trend and a negative a downward one, respectively.

Table A1-3: Mann Kendal Test - Z and p values, 1978-2008

7-700 t		> 700 t		> 7 t	
p	Z	p	Z	p	Z
0.0000	-5.5494	0.000230	-4.0755	0.0000	-5.4300

The results of the Mann-Kendall test (under the alternative hypothesis) at a 95% confidence level display negative Z values. In plain English this means that there is enough evidence for the existence of a downward trend in all three categories for the period 1978-2008 (see Table A-1-3). Furthermore, all p -values are close to zero, which means that the trend is not caused by random sampling.

Appendix A2-1 Initial Regression Analysis of Cleanup Cost

The equation of the initial fitted model using linear regression is :

$$\text{LOG10}(\text{CleanupCost}) = 4.73714 + 0.612415 * \text{LOG10}(V)$$

Coefficients

	<i>Least Squares</i>	<i>Standard</i>	<i>T</i>	
<i>Parameter</i>	<i>Estimate</i>	<i>Error</i>	<i>Statistic</i>	<i>P-Value</i>
Intercept	4.73714	0.142437	33.2579	0.0000
Slope	0.612415	0.0579829	10.562	0.0000

Analysis of Variance

<i>Source</i>	<i>Sum of Squares</i>	<i>Df</i>	<i>Mean Square</i>	<i>F-Ratio</i>	<i>P-Value</i>
Model	51.0298	1	51.0298	111.56	0.0000
Residual	37.9674	83	0.457439		
Total (Corr.)	88.9972	84			

Correlation Coefficient = 0.757223

R-squared = 57.3386 percent

R-squared (adjusted for d.f.) = 56.8247 percent

Standard Error of Est. = 0.676342

Mean absolute error = 0.521885

Durbin-Watson statistic = 1.51211 (P=0.0107)

Lag 1 residual autocorrelation = 0.240161

The R-Squared statistic indicates that the model as fitted explains 57.3386% of the variability in LOG10(CleanupCost). The correlation coefficient (Pearson’s correlation coefficient p) equals 0.757223, indicating a strong relationship between the variables. Both of these values indicate that the model fit is satisfactory.

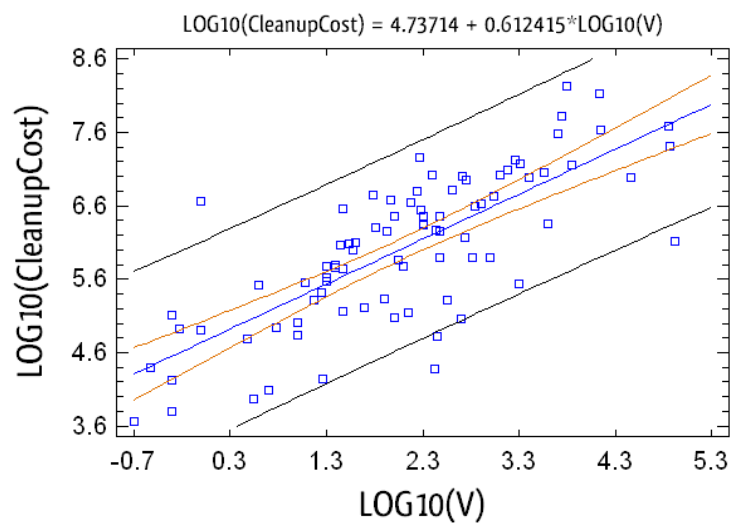


Figure A2-1: Linear Regression of Log(Spill Size) and Log (Cleanup Cost) – 95% confidence

To check whether the y-values come from a normal distribution an analysis of the residuals has been performed.

We also performed an analysis of variances (ANOVA) which indicated that there is a statistically significant relationship between LOG10(CleanupCost) and LOG10(V) at the 95.0% confidence level.

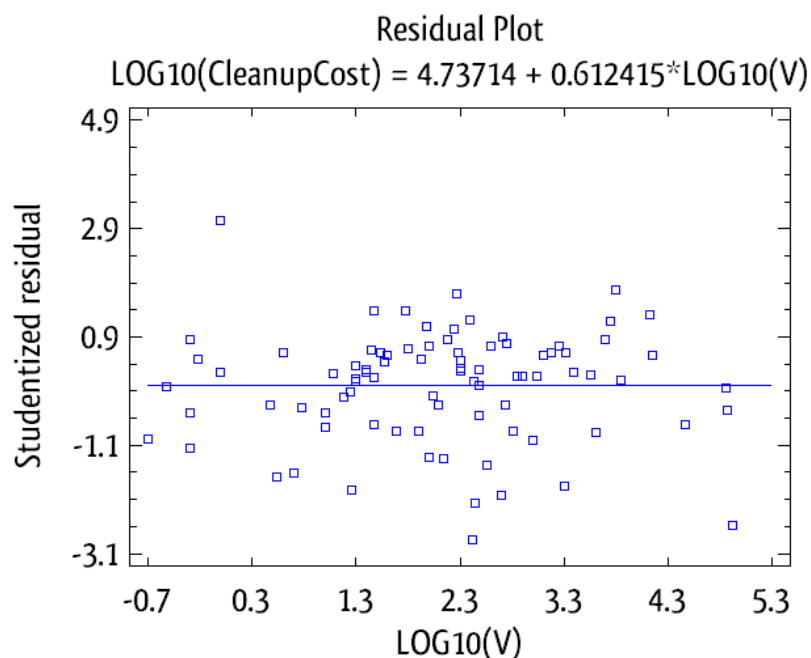


Figure A2-2: Residual Plot of the Linear Regression of Log(Spill Size) and Log (Cleanup Cost) The standard deviation of the residuals (standard error of the estimate) is 0.676342 and the average value (mean absolute error, MAE) is 0.521885. The Durbin-Watson (DW) statistic tests the residuals to determine if there is any significant correlation based on the order in which they occur in your data file. Since the P-value is less than 0.05, there is an indication of possible serial correlation at the 95.0% confidence level.

Furthermore, there is a strong indication that there is an outlier in the data since the studentized residual is greater than 3. This refers to the Daiwa Maru No 18 spill of just 1 tonne that happened in 1997 in Kawasaki, Japan due to mishandling of oil supply. This spill had a cleanup cost of about 416 million Yens or about 4.6 million 2008 US \$. This is also the per tonne clean up cost. This value is indeed an outlier.

Table A2-1: Unusual Residual of the Linear Fit

			<i>Predicted</i>		<i>Studentized</i>
<i>Row</i>	<i>X</i>	<i>Y</i>	<i>Y</i>	<i>Residual</i>	<i>Residual</i>
40	4.92428	6.12301	7.75284	-1.62983	-2.58
41	2.69897	5.05316	6.39003	-1.33687	-2.03
63	0.0	6.66377	4.73714	1.92663	3.06
68	2.44716	4.81712	6.23582	-1.4187	-2.16
70	2.4183	4.38248	6.21814	-1.83567	-2.85

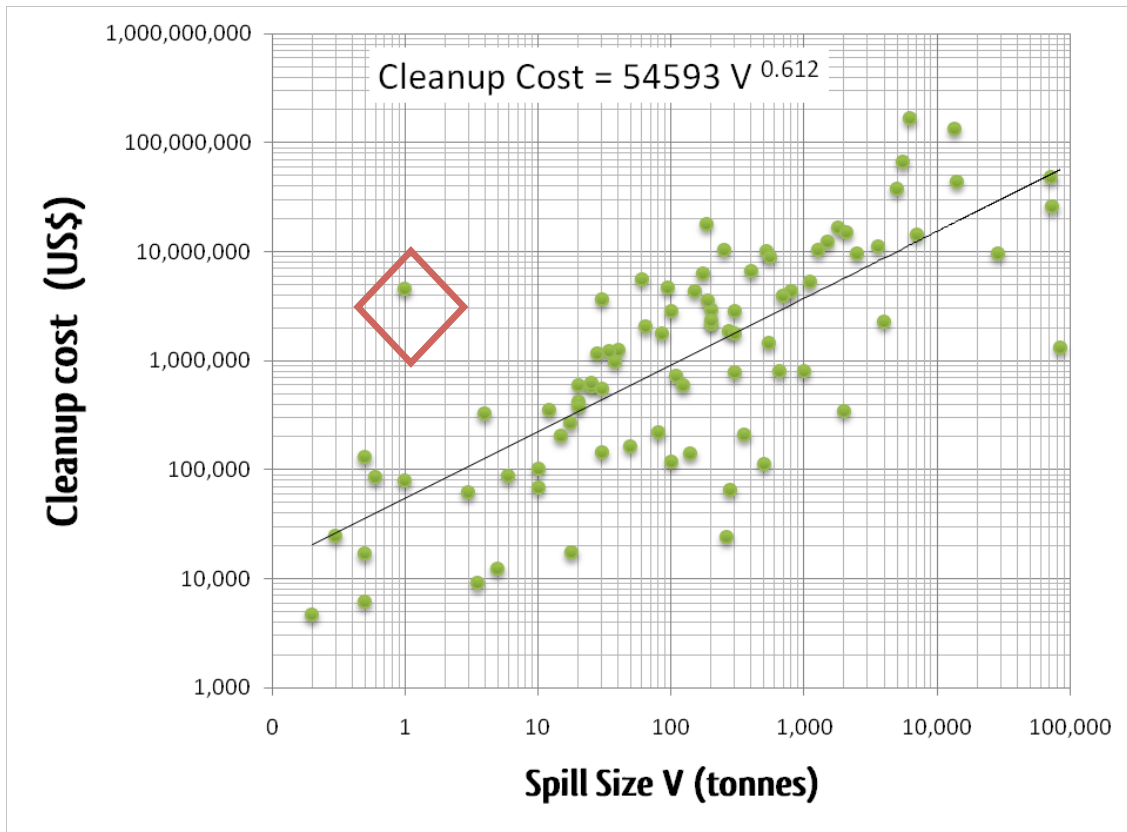


Figure A2-3: Initial Linear Regression of Log(Spill Size) and Log (Cleanup Cost)

Appendix A2-2 Initial Regression Analysis of Total Cost

The equation of the initial fitted model using linear regression is :

$$\log_{10}(\text{Total Cost}) = 4.84722 + 0.652006 * \log_{10}(V)$$

Coefficients

	<i>Least Squares</i>	<i>Standard</i>	<i>T</i>		<i>M.A.D.</i>
<i>Parameter</i>	<i>Estimate</i>	<i>Error</i>	<i>Statistic</i>	<i>P-Value</i>	<i>Estimate</i>
Intercept	4.84722	0.144616	33.5178	0.0000	4.82202
Slope	0.652006	0.0577042	11.2991	0.0000	0.695397

Analysis of Variance

<i>Source</i>	<i>Sum of Squares</i>	<i>Df</i>	<i>Mean Square</i>	<i>F-Ratio</i>	<i>P-Value</i>
Model	69.6119	1	69.6119	127.67	0.0000
Residual	48.5273	89	0.54525		
Total (Corr.)	118.139	90			

Correlation Coefficient = 0.767617

R-squared = 58.9236 percent

R-squared (adjusted for d.f.) = 58.4621 percent

Standard Error of Est. = 0.73841

Mean absolute error = 0.535375

Durbin-Watson statistic = 1.73303 (P=0.1008)

Lag 1 residual autocorrelation = 0.104609

Mean absolute deviation = 0.531643

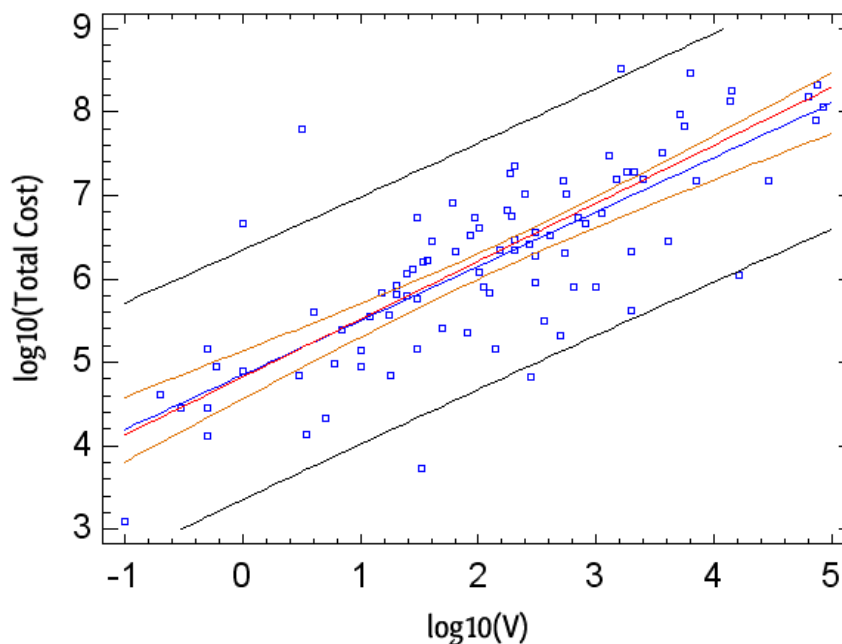


Figure A2-4: Linear Regression of Log(Spill Size) and Log (Cleanup Cost) – 95% confidence

CAUSES, TRENDS OF OIL SPILLS AND REGRESSION ANALYSIS OF IOPCF COST DATA

The R-Squared statistic indicates that the model as fitted explains 58.9236% of the variability in LOG10(TotalCost). The correlation coefficient (Pearson's correlation coefficient r) equals 0.767617, indicating a strong relationship between the variables. Both of these values indicate that the model fit is satisfactory.

To be mathematically consistent one should check all assumptions of linear regression.

The two necessary conditions that must be met in a linear model ($Y=aX+b$) is that the Y's should have a normal distribution for each value of x.

There must be a constant amount of spread (standard deviation) for each value of x. (homoscedasticity condition)

To check whether the y-values come from a normal distribution an analysis of the residuals has been performed.

We also performed an analysis of variances (ANOVA) which indicated that there is a statistically significant relationship between LOG10(CleanupCost) and LOG10(V) at the 95.0% confidence level.

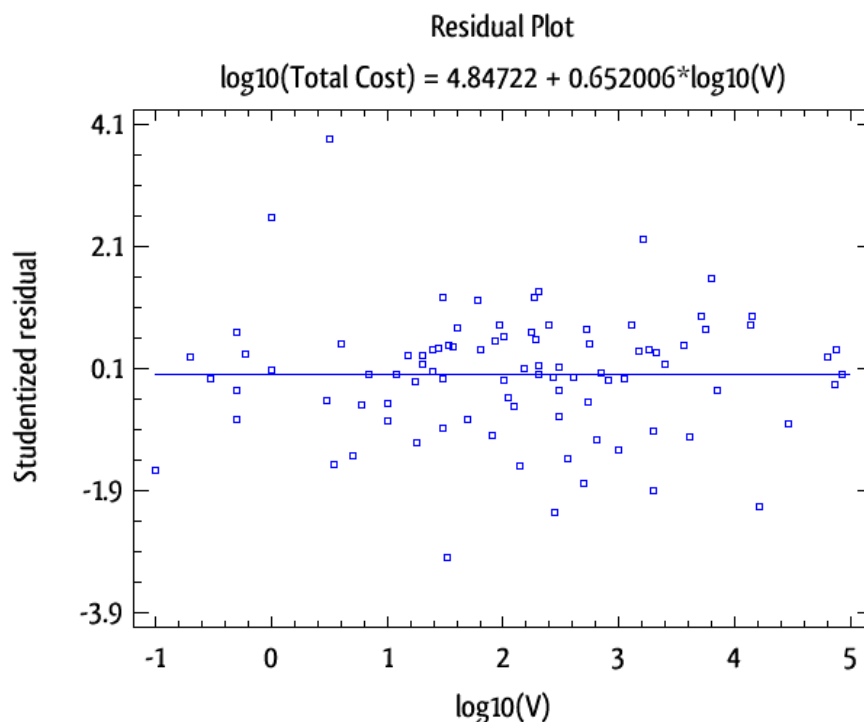


Figure A2-5: Residual Plot of the Linear Regression of Log(Spill Size) and Log (Cleanup Cost)

The mean absolute error (MAE) of 0.535375 is the average value of the residuals. The Durbin-Watson (DW) statistic tests the residuals to determine if there is any significant correlation based on the order in which they occur in your data file. Since the P-value is greater than 0.05, there is no indication of serial autocorrelation in the residuals at the 95.0% confidence level.

Furthermore, there is a strong indication that there is are outlier in the data since the studenized residual is greater than 3.

Table A2-2: Unusual Residual of the Linear Fit

Row	X	Y	Predicted Y	Residual	Studentized Residual	Resistant Model Residual
8	4.20412	6.04489	7.58833	-1.54344	-2.17	-1.70066
12	1.51851	3.72922	5.8373	-2.10808	-3.00	-2.14877
65	0.50515	7.79994	5.17658	2.62336	3.87	2.62665
70	0.0	6.66467	4.84722	1.81745	2.59	1.84265
74	2.44716	4.81712	6.44278	-1.62567	-2.27	-1.70664
91	3.20412	8.52248	6.93633	1.58615	2.21	1.47233

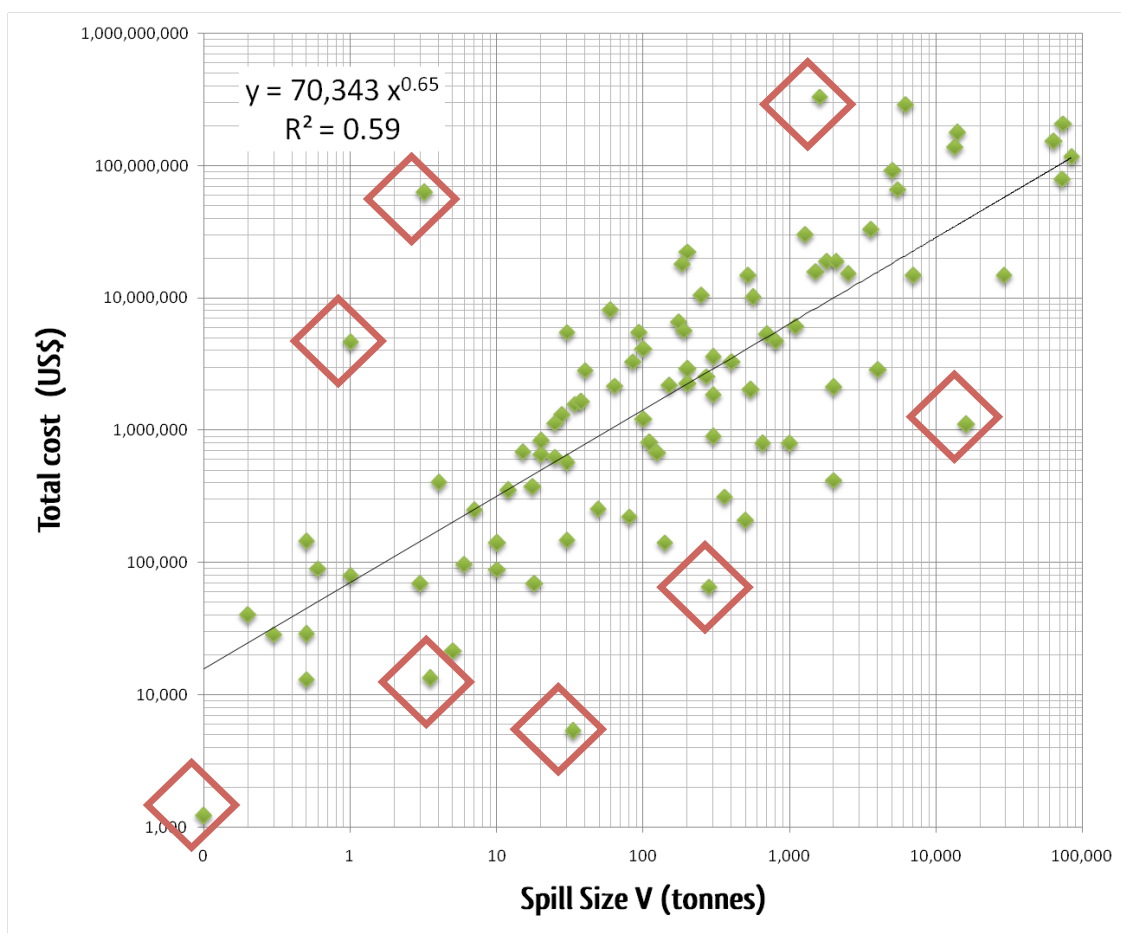


Figure A2-6: Initial Linear Regression of Log(Spill Size) and Log (Total Cost)

B

DRAFT AMENDMENT TO FSA GUIDELINES – ENVIRONMENTAL RISK EVALUATION CRITERIA

The author of this dissertation has been involved in various IMO submissions on the issue of Formal Safety Assessment, see list below.

List of submissions to the IMO

Marine Safety Committee (MSC)

1. "Application of the FSA Guidelines and review of FSA studies", MSC 86/17/2
2. "FORMAL SAFETY ASSESSMENT: Possible Improvements on FSA Guidelines", MSC 82/INF.3, (Nov. 29-Dec. 8, 2006)

Marine Environment Protection Committee (MEPC)

3. "Formal Safety Assessment - Environmental Risk Evaluation Criteria", MEPC 56/18/1, (May, 2007)
4. "An Empirical Analysis of IOPCF Oil Spill Cost Data" (Abridged Ver.), MEPC 60/17 - Annex 2 (July, 2009)
5. "Further experience with non-linear oil spill cost functions", MEPC 61/18/2 (July, 2010)
6. "Combining environmental and safety criteria and selection of a severity matrix", MEPC62/18
7. "Further experience with oil spill databases and update of non-linear oil spill cost functions", MEPC62/18/1 (July 2011)

Specifically, he has been involved in raising the issue of environmental risk evaluation criteria (MEPC 56/18/1) and in the working group that drafted the amendment to FSA Guidelines (MEPC 62/WP.13) which is attached below.

MARINE ENVIRONMENT PROTECTION
COMMITTEE
62nd session
Agenda item 18

MEPC 62/WP.13
13 July 2011
Original: ENGLISH

DISCLAIMER

As at its date of issue, this document, in whole or in part, is subject to consideration by the IMO organ to which it has been submitted. Accordingly, its contents are subject to approval and amendment of a substantive and drafting nature, which may be agreed after that date.

FORMAL SAFETY ASSESSMENT

Report of the Working Group on Environmental Risk Evaluation Criteria within the context of Formal Safety Assessment

Introduction

1 The Working Group on Environmental Risk Evaluation Criteria within the context of Formal Safety Assessment met from 11 to 13 July 2011, under the Chairmanship of Professor Harilaos N. Psaraftis (Greece).

2 The Group was attended by delegations from:

ANTIGUA AND BARBUDA
BRAZIL
CHINA
DENMARK
GERMANY
GREECE
JAPAN

MARSHALL ISLANDS
NORWAY
REPUBLIC OF KOREA
RUSSIAN FEDERATION
SPAIN
UNITED KINGDOM
UNITED STATES

an observer from the following intergovernmental organization:

INTERNATIONAL OIL POLLUTION COMPENSATION FUNDS (IOPCF)

and by observers from the following non-governmental organizations in consultative status:

INTERNATIONAL ASSOCIATION OF CLASSIFICATION SOCIETIES (IACS)
OIL COMPANIES INTERNATIONAL MARINE FORUM (OCIMF)
INTERNATIONAL ASSOCIATION OF INDEPENDENT TANKER OWNERS
(INTERTANKO)
CRUISE LINES INTERNATIONAL ASSOCIATION (CLIA)
THE ROYAL INSTITUTION OF NAVAL ARCHITECTS (RINA)
INTERFERRY

Terms of Reference

3 Using documents MEPC 62/18, MEPC 62/18/1, MEPC 62/18/2, MEPC 62/18/3, MEPC 62/18/4 and MEPC 62/INF.24 as a basis, and any other relevant information, the Working Group was instructed to:

- .1 finalize, in Step 4 of the FSA, an appropriate volume-dependent CATS global threshold scale or function for ascertaining if a specific Risk Control Option (RCO) is cost-effective, including its integration within the FSA methodology;
- .2 recommend a way of combining environmental and safety criteria for those RCOs that affect both environmental and fatality risk;
- .3 conclude on an appropriate risk matrix or index for environmental criteria;
- .4 recommend an appropriate ALARP region and F-N diagram, including an appropriate value for the slope of the F-N curve; and
- .5 submit a written report to plenary on Thursday, 14 July 2011.

4 Documents MEPC 62/18/1, MEPC 62/18/2, MEPC 62/18/3, MEPC 62/18/4 and MEPC 62/INF.24 were first introduced by the originators and commented upon by the Group. Document MEPC 62/18 was introduced and discussed later, in the context of Terms of Reference 3 and 4.

Finalize, in Step 4 of the FSA, an appropriate volume-dependent CATS global threshold scale or function for ascertaining if a specific Risk Control Option (RCO) is cost-effective, including its integration within the FSA methodology

5 As agreed in previous meetings, the costs associated with an oil spill are not directly proportional to the volume spilled (i.e. the cost per unit volume spilled is not constant). Therefore, the Group concluded that in general, it is not possible to define appropriate cost effectiveness criteria based on CATS (Cost to Avert a Tonne of Oil Spilled). The Group concluded that the approach to be followed to ascertain if a specific RCO is cost-effective should be primarily based on a cost-benefit approach, i.e. on whether the costs to implement the RCO are less than the risk reduction introduced, followed by an appropriate analysis of its cost-effectiveness. To evaluate this risk reduction, it is necessary to convert volume of oil spilled into costs associated with it. Several proposals of a volume-dependent function have been submitted to MEPC over the past years, with no agreement on any specific one.

6 Noting that the most appropriate conversion formula to use will depend on the specific scope of each FSA to be performed, the Group decided to outline a general approach to be followed, rather than prescribe a unique conversion formula to use.

7 With this objective in mind, the Group noted that the basis of this work essentially consists of:

- .1 the consolidated oil spill database;
- .2 the regression analyses;
- .3 the relevant MEPC 62 submissions; and
- .4 the FSA Guidelines.

8 After some discussion, the Group agreed on the approach outlined in the subsequent paragraphs.

9 The consolidated oil spill database is described in document MEPC 62/INF.24 and is based on:

- .1 IOPCF data;
- .2 US Data; and
- .3 Norwegian data.

10 The Group agreed that the consolidated oil spill database should be in the public domain and maintained at a website (IMO or other).

11 The Group acknowledged that the consolidated oil spill database was probably the best database available within the time limits and resources available to the four delegations that worked on it (Germany, Greece, Japan and the United States). However, its limitations and possible deficiencies should be acknowledged. These are described in document MEPC 62/INF.24 and may also involve incomplete or missing data on costs or other information.

12 Depending on the subset of the consolidated oil spill database under consideration, the Group agreed that the following volume-based total spill cost functions could be appropriate to be used in environmental FSA studies:

Dataset	Total Spill Cost (2009 US dollars)	Reference
All spills	$67,275 V^{0.5893}$	MEPC 62/INF.24
V>0.1 tonnes	$42,301 V^{0.7233}$	MEPC 62/18 ¹

Where V is spill size in tonnes.

13 The Group agreed that the oil spill database used in an environmental FSA is subject to updating as new spill cost data become available, from both the above sources and elsewhere, and that FSA analysts are free to perform new regression analyses, as long as these are well documented.

14 In the discussion on an appropriate "assurance factor", reflecting society's willingness to pay to avert an oil spill instead of waiting for the accident and incurring its damages, some delegations suggested that its value should not be the subject of an FSA but should be left open for policy makers to decide. Other delegations proposed that its value should be the subject of an FSA, and the example of a value of 1.5 for project SAFEDOR was suggested as an example. The Group agreed that any assurance factor in an environmental FSA should also be well documented.

15 In the discussion on an appropriate "uncertainty factor", reflecting possible uncertainties in the spill cost or other data, the Group agreed that including an uncertainty factor should be part of an environmental FSA and should also be well documented.

16 Appropriate values for the assurance factor and the uncertainty factor should be derived for use in the FSA.

17 With respect to the inclusion of the uncertainty factor, one delegation did not share this view, arguing that if any uncertainty in the estimation of the real cost of an oil spill can be computed, this can be reflected in an update of the relevant cost information without the need of such a factor. This delegation also argued that the spill costs of the consolidated database actually overestimate real costs due to the fact that, from an accounting

¹ Updated regression made on the final consolidated dataset.

perspective, they are accounted for at the year of the accident and not at the year of payment, which may be as many as 15-20 years later. This leads to an overestimation in the NPV of the spill cost, making the need for an uncertainty factor unnecessary and rendering the analysis conservative.

18 Some delegations indicated that the third column in figure 1 of document MEPC 62/18/3, reflecting claimed costs, is not necessarily lower than the first column (real total costs), as the figure suggests. That this may be the case was confirmed by the IOPCF representative, according to whom claims can be very high at times but real costs for the IOPCF represent settled costs.

19 The Group noted that because of the above possibility, the uncertainty factor may be less than one in some cases, to the extent that this can be shown.

20 The Group discussed the possibility of conceivably working with a regression line, not at 50%, but at a different percentile.

21 As regards the application of a non-linear cost function within FSA, the Group noted the relevant work of (among others) Yamada (2009)², Kontovas *et al.* (2010)³, Hamann and Loer (2010)⁴ and Yamada and Kaneko (2010)⁵, documents MEPC 59/17/1 (Japan) and MEPC 62/18/2. The Group agreed that the user should be free to develop new approaches taking into account the objectives of the FSA, provided these are well documented.

22 The Group also agreed that each environmental FSA must address all aspects of the cost evaluation.

Recommend a way of combining environmental and safety criteria for those RCOs that affect both environmental and fatality risk

23 Greece introduced document MEPC 62/18, proposing an approach for an RCO that affects both environmental and fatality risk.

24 After discussion, the Group agreed to use the approach outlined in section 4.2 of the annex to this document (see paragraph 33 below).

25 The Group also emphasized that all cost and benefit components of the cost-benefit or cost-effectiveness inequality should be shown in an FSA study for better transparency.

Conclusion on an appropriate risk matrix or index for environmental criteria

26 During MEPC 60, the Working Group had proposed the Frequency Matrix already established for FSA methodology (MSC 83/INF.2) for consideration by the Committee. The MEPC 60 Group had further agreed that the severity index needs to be consistent with the CATS function, but until such time that the CATS function is resolved, the exact detail of the severity index cannot be finalized.

² Yamada, Y., 2009. The Cost of Oil Spills from Tankers in Relation to Weight of Spilled Oil, *Marine Technology*, 46(4). pp. 219-228.

³ Kontovas, C.A., Psaraftis, H.N., Ventikos N. "An Empirical Analysis of IOPCF Oil Spill Cost Data", *Marine Pollution Bulletin*, 60(9), pp. 1455-1466.

⁴ Hamann, R. and Loer, K., 2010. Risk-based Optimisation of Crude Oil Tanker Cargo Holds. Proc. of The 3rd International Symposium on Ship Operations, Management & Economics, Athens, Greece 7-8 October, 2010.

⁵ Yamada, Y. and Kaneko, F., 2010. On the Derivation of CATSthr within the Framework of IMO environmental FSA studies. Proc. of the 5th International Conference on Collision and Grounding of Ships, June 14th - 16th 2010, Espoo, Finland.

27 The Group considered a relevant proposal on this subject, document MEPC 61/18/2 (Greece), recommending a volume-based severity index. Some delegations raised concerns that the proposed approach implied a linear relationship between spill size and consequence and recommended instead a severity index based on monetary damages (US dollars). However, after some discussion, a majority of the members of the Group agreed on a volume-based severity index.

28 The Group subsequently agreed to modify the severity index labels into "category 1, 2, 3, etc.", instead of "minor, significant, severe, catastrophic and disastrous" so as not to associate them with equivalent labels in the severity matrix used in the safety FSA (fatalities and injuries). The Group also agreed to add an additional category of spill size below one tonne. The finalized severity matrix is shown below and constitutes an example.

Severity Index		
SI	SEVERITY	DEFINITION
1	Category 1	Oil spill size < 1 tonne
2	Category 2	Oil spill size between 1-10 tonnes
3	Category 3	Oil spill size between 10-100 tonnes
4	Category 4	Oil spill size between 100-1,000 tonnes
5	Category 5	Oil spill size between 1,000-10,000 tonnes
6	Category 6	Oil spill size >10,000 tonnes

29 At the same time, the Group reiterated that, in using a volume-based severity index in the HAZID step, this does not mean that spill size is the only determinant of the severity of an oil spill accident. In that regard, the Group recalled extensive discussions within the relevant MEPC correspondence groups on this subject over the last four years (see for instance documents MEPC 57/17 and MEPC 58/17), to the effect that, in addition to spill size, there are a number of other parameters that can have a significant impact on the severity of an oil spill, such as oil type, location, weather conditions, season, geomorphology of the shoreline, and others. The scatter of diagrams plotting spill cost versus spill size (for instance figure 5 of document MEPC 62/INF.24) can be explained by the existence of these parameters. Such parameters should be given due consideration where appropriate.

Recommendations on an appropriate ALARP region and F-N diagram, including an appropriate value for the slope of the F-N curve

30 The Working Group at MEPC 60 had noted that it was premature to take the issue further until such time that the CATS function had been resolved.

31 As there were no new submissions on this matter to MEPC 62, the Group recalled several useful previous submissions, notably by Germany (MEPC 59/17) and by Japan (MEPC 60/17/3).

32 The Group questioned the usefulness of F-N diagrams within an environmental FSA, particularly given the perception of the Group that the current fleet is in the ALARP region.

33 After some discussion, and given the lack of data or new submissions on the subject, the Group agreed that this matter should be left open until FSA studies that have an environmental dimension have an opportunity to perform an analysis and present some results.

Other relevant issues and consideration on how to complete the work

34 On the basis of all previous paragraphs, the Group, after input from a splinter group (subgroup) tasked to work on the subject, agreed to prepare a document as an annex to this

document. After thorough review and discussion by the Group, it was agreed to request the Committee to forward this annex to the Maritime Safety Committee for inclusion in the FSA Guidelines.

35 The delegations of Brazil and China requested the following statement to be included:

"Brazil and China have concerns about the establishment of a subgroup within the Working Group, especially when the draft report was already prepared and only subject to minor editorial changes. It is also our view that the Terms of Reference were not clear enough to ask the Group to prepare a specific text which could be directly incorporated into the FSA Guidelines. In view of this, Brazil and China reserve their position."

36 The Group, with the exception of Brazil, considered that its deliberations sufficiently fulfilled the Terms of Reference and that, after four years of work on the subject, which involved numerous discussions on a wide array of topics in which divergence of opinion was very common, the Group was finally able to overcome the difficulties and converge on several subjects. In that sense, it is the Group's opinion that a significant milestone has been achieved and that the Committee should be able to report the outcome of this work to the Maritime Safety Committee for consideration to include the relevant parts into the IMO FSA Guidelines.

37 The Group also expressed the opinion that environmental dimensions other than oil pollution may warrant similar consideration within FSA, and that the methodology used herein could be extended and/or adapted to cover these other dimensions, in case this is deemed appropriate by the Organization.

Action requested of the Committee

38 The Committee is invited to approve the report in general, and, in particular, to:

- .1 endorse the consolidated oil spill database and request the Secretariat to arrange to make it publicly available (paragraphs 9 to 11);
- .2 endorse the Group's approach on an appropriate volume-dependent CATS global threshold function for ascertaining if a specific Risk Control Option (RCO) is cost-effective, including the cost functions proposed and its integration within the FSA methodology (paragraphs 11 to 22, and the annex);
- .3 endorse the Group's proposal on how to combine environmental and safety criteria for RCOs that reduce environmental and safety risk (paragraphs 28 to 29);
- .4 endorse the Group's proposal on how to proceed on the ALARP region and F-N curves (paragraph 33); and
- .5 invite the Maritime Safety Committee to consider the outcome of the work on environmental risk evaluation criteria, and in particular, the annex to this document, for inclusion into the FSA Guidelines.

ANNEX

FSA ENVIRONMENTAL RISK EVALUATION CRITERIA
DRAFT AMENDMENT TO FSA GUIDELINES
A PROPOSED NEW APPENDIX

Noting that the most appropriate conversion formula to use will depend on the specific scope of each FSA to be performed, a general approach to be followed is outlined in the following suggested examples.

1. Consolidated oil spill database based on:
 - a. IOPCF data;
 - b. US Data;
 - c. Norwegian data;

Figure 1 shows the data of the consolidated oil spill database in terms of specific costs per tonne spilled (Figure 5 of document MEPC 62/INF.24). Further information with respect to the basis of the database can be found in document MEPC 62/INF.24. It should be acknowledged that the consolidated oil spill database has limitations and possible deficiencies. These are described in document MEPC 62/INF.24 and may also involve incomplete or missing data on costs or other information.

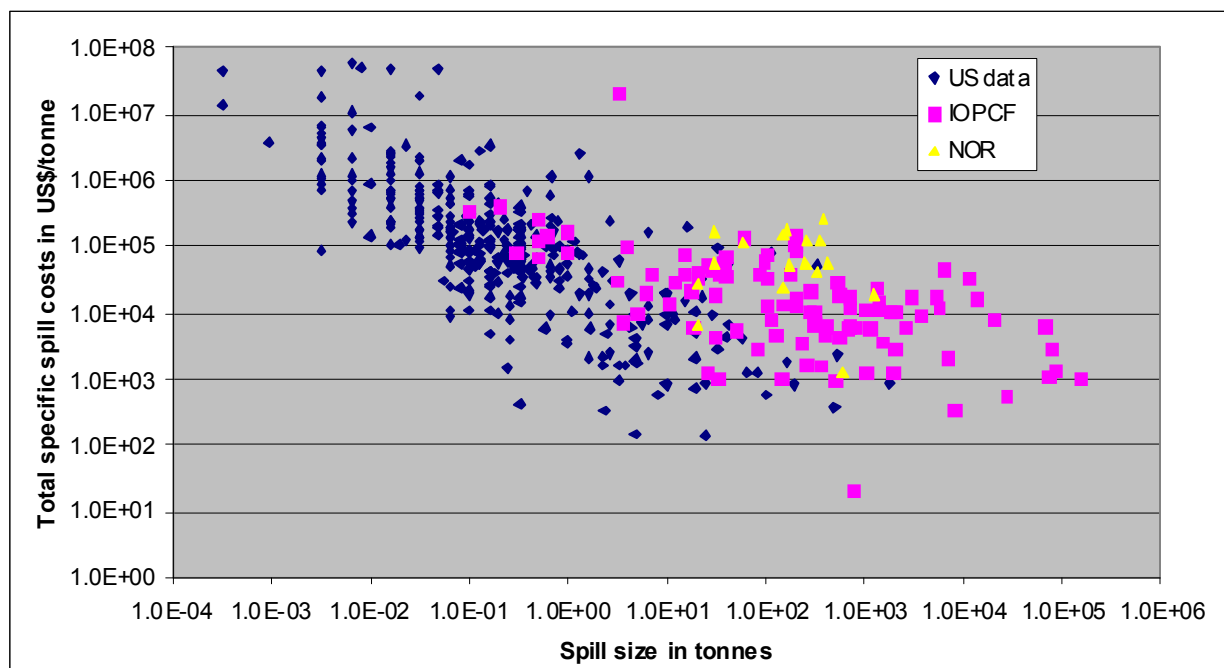


Figure 1: All specific oil spill cost data in 2009 USD (spill cost per tonne). Source: document MEPC 62/INF.24.

The submitter of the FSA can amend this database with new oil spill data, however, this amendment should be properly documented.

2. Some regression formulae derived from the consolidated oil spill database are summarized in Table 1 in which V is spill size in tonnes.

Table 1: Regression formulae derived from the consolidated database

Dataset	f(V)=Total Spill Cost (TSC) (2009 US dollars)	Reference
All spills	67,275 V ^{0.5893}	MEPC 62/INF.24
V>0.1 tonnes	42,301 V ^{0.7233}	MEPC 62/18 ⁶

FSA analysts are free to use other conversion formulae, so long as these are well documented by the data. For example, if an FSA is considering only small spills, the submitter may filter the data and perform his or her own regression analysis.

3. It is recommended that the FSA analyst use the following formula to estimate the societal oil spill costs (SC) used in the analysis:

$$SC_{threshold} = F_{Assurance} * F_{Uncertainty} * f(V)$$

This equation considers:

1. Assurance factor ($F_{Assurance}$): allowing for society's willingness to pay to avert accidents;
2. Uncertainty factor ($F_{Uncertainty}$): allowing for uncertainties in the cost information from occurred spill accidents; and
3. Volume-dependent total cost function ($f(V)$): representing the fact that the cost per unit oil spilled decreases with the spill size in US\$ per tonne oil spilled.

The values of both assurance and uncertainty factors should be well documented.

In order to consider the large scatter, the FSA analyst may perform a regression to determine a function $f(V)$ that covers a percentile different than 50 % and document it in the report.

4. Application in RCO evaluation

The FSA analyst should perform a cost-benefit and cost-effectiveness evaluation of the RCOs identified and provide all relevant details in the report, as outlined below.

4.1 RCOs affecting oil spills only

In case an RCO affects oil spills only:

RCO is cost effective if $\Delta C < \Delta SC$

ΔC = Expected cost of the RCO,

ΔSC = (Expected SC **without** the RCO) – (Expected SC **with** the RCO) = Expected benefit of the RCO.

⁶ Updated regression made on the final consolidated dataset.

4.2 RCOs affecting both safety and environment

In case of RCOs addressing both safety and environment the following formula is recommended:

$$NCAF = (\Delta C - \Delta SC) / \Delta PLL$$

In the above,

ΔC = Expected cost of the RCO
 ΔSC = (Expected SC **without** the RCO) – (Expected SC **with** the RCO) = Expected benefit of the RCO,
 ΔPLL = Expected reduction of fatalities due to the RCO, and

The criteria for NCAF are as per table 2 of appendix 7 of document MSC 83/INF.2.

In case there is an economic benefit (ΔB), ΔC should be replaced by $\Delta C - \Delta B$.

It is also emphasized that all cost and benefit components of the cost-benefit or cost-effectiveness inequality should be shown in an FSA study for better transparency.

The user is free to develop new approaches taking into account the objectives of the FSA.

5. Index for environmental criteria

The table below should be inserted **in Appendix 4 (paragraph 3) of the FSA Guidelines**

Severity Index		
SI	SEVERITY	DEFINITION
1	Category 1	Oil spill size < 1 tonne
2	Category 2	Oil spill size between 1-10 tonnes
3	Category 3	Oil spill size between 10-100 tonnes
4	Category 4	Oil spill size between 100-1,000 tonnes
5	Category 5	Oil spill size between 1,000-10,000 tonnes
6	Category 6	Oil spill size >10,000 tonnes

Curriculum Vitae

Personal information

First name(s) / Surname(s)

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Nationality

Greek

Date of birth

22 September 1979

Gender

Male

Work experience

Dates

01 March 2007

Occupation or position held

Doctoral Research

Name and address of employer

Laboratory for Maritime Transport, National Technical University of Athens
9, Iroon Politechneiou Str., GR-15771 Zografou (Greece)

Type of business or sector

Academia

Dates

01 September 2000 - 01 September 2002

Occupation or position held

Software Developer

Main activities and responsibilities

Designing e-learning material for scientific computing (e.g. Matlab and Mathematica)

Name and address of employer

Central Computer Center, National Technical University of Athens
9, Iroon Politechneiou, GR-15771 Zografou (Greece)

Type of business or sector

Academia

Education and training

Dates

June 2007 – December 2011

Title of qualification awarded

Phd

Principal subjects covered

Risk Assessment for maritime safety and environmental protection (oil spills and air emissions)

Name and type of organisation providing education and training

School of Naval Architecture
National technical University of Athens, GR

Level in national or international classification

PhD Student - Technical University

Dates

01 May 2006 - February 2007

occupational skills covered

Engineering

Name and type of organisation providing education and training

Greek Army Corps of Engineers

Level in national or international classification

Military Service



Dates 01 September 1999 - 01 September 2005
 Title of qualification awarded **Diploma of Engineering**
 Principal subjects **Naval Architecture and Marine Engineering**
 Name and type of organisation providing education and training School of Naval Architecture and Marine Engineering
 National Technical University of Athens
 Level in national or international classification Technical University

Dates September 1998 - September 1999
 Title of qualification awarded selected courses
 Principal subjects covered Mathematics
 Name and type of organisation providing education and training **Department of Mathematics**
University of Athens, GR
 Level in national or international classification University

Personal skills and competences

Mother tongue(s) **Greek**

Other language(s)

Self-assessment
 European level (*)

English
German

Understanding				Speaking				Writing	
Listening		Reading		Spoken interaction		Spoken production			
C1	Proficient user	C1	Proficient user	C1	Proficient user	C1	Proficient user	B2	Independent user
B1	Independent user	B2	Independent user	A2	Basic User	A2	Basic User	B1	Independent user

(*) [Common European Framework of Reference \(CEF\) level](#)

Organisational skills and competences

- Society of Naval Architects and Marine Engineers (SNAME)
 Chairman of the NTUA Student Section [Mar. 2009-Mar. 2010, Mar. 2010 – Apr. 2011]
- Board of European Student Organization (BEST)
 Vice-President of the local group of Athens (LBG Athens) [2002-2003]

Member of

- Technical Chamber of Greece (Greece)
- Society of Naval Architects and Marine Engineers (USA)
- Society for Risk Analysis (USA)
- International Association of Maritime Economists (IAME)
- Hellenic Association of Maritime Economists (ENOE)

Computer skills and competences

- Operating Systems (MS Windows and linux-based mainly Debian and Ubuntu).
- Office Suites (Word processing, Presentations, Spreadsheets): All major ones (MS Office, OpenOffice, Libre Office)
- Programming Skills : HTML, Fortran, Pascal
- Web authoring: webmasteHTML,CSS coding and IIS and Apache
- Image Processing: Adobe Photoshop CS, Corel PaintShop Pro, GIMP
- CAD: Autocad, Rhinoceros
- Marine Design: Autoship, Tribon/AVEVA, Maxsurf
- Risk And Decision Analysis Software : Palisade @Risk, Crystal Ball

Artistic skills and competences

Technical drawing and Painting (pencil, oil and acrylics)

Driving licence(s) B

Additional information (major and non-confidential projects only)

- "Ship Air Emissions Study", Study funded by Hellenic Chamber of Shipping (January- June 2008) Modelling and Estimating world CO2 emissions inventory for shipping
- "Hazard identification (Hazid) of Tanker Operations", Task 4.7, FSA on crude oil carriers within Project SAFEDOR (Design, operation and regulation for safety), project funded by the 6th framework programme of the European Commission.
- "Effective Bulk Transport," Gift Agreement funded by Det Norske Veritas (DNV) (January 2008-December 2010).
- "Assessment of Environmental Impact in Marine Transportation and Related Activities," project funded by American Bureau of Shipping (ABS) (June 2008-May 2011).
- "Centre of Excellence in Ship Total Energy-Emissions-Economy", a centre established by the Lloyd's Register Educational Trust (LRET)
- EU DG Climate Action Study : "Study on ships visiting EU ports", CLIMA.B.3/ETU/2010/0021rl, Principal contractor: IHS Global Limited (2010)

Involvement at the International Maritime Organization

- Assisting the LMT team that is active at the IMO, both at the Maritime Safety Committee (MSC) and at the Marine Environment Protection Committee (MEPC).
 - *Member of the Greek delegation at MEPC 62.*
 - *Areas involved: Formal Safety Assessment within MSC and Environmental Risk Evaluation Criteria and GHG emissions from ships within MEPC.*

Annexes **Scientific Publications**

For a complete list visit: <http://users.ntua.gr/kontovas/pub.html>

Total of peer reviewed publications (2007-2011): 21 (journals:7. Book chapters: 2, conf. proc.:12)

REFERRED JOURNALS

2009

- Kontovas, C.A., Psaraftis, H.N. "Formal Safety Assessment: A critical review", Marine Technology, Vol. 46, No 1, January 2009, [\[download\]](#)
- Psaraftis, H.N., Kontovas, C.A. "CO2 Emissions Statistics for the World Commercial Fleet", WMU Journal of Maritime Affairs", Vol. 8 (2009), No.1, 1-25 [\[download\]](#)
- Psaraftis, H.N., Kontovas, C.A. "A web-based open emissions calculator", Int. J. Ocean Systems Management, Vol. 1, No. 2, 2009 [\[download\]](#)

2010

- Psaraftis, H.N., Kontovas, C.A. "Balancing the Economic and Environmental Performance of Maritime Transportation", Transportation Research Part D: Transport and Environment, Volume 15, Issue 8, December 2010, Pages 458-462 [\[download\]](#)
- Kontovas, C.A., Psaraftis, H.N., Ventikos N. "An Empirical Analysis of IOPCF Oil Spill Cost Data", Marine Pollution Bulletin, Volume 60, Issue 9, September 2010, Pages 1455-1466 [\[download\]](#)

2011

- Kontovas, C.A., Psaraftis, H.N. (2011), "Reduction of Emissions Along the Maritime Intermodal Container Chain", Maritime Policy & Management, Volume 38, No. 4, pp 455-473, July 2011 [\[download\]](#)
- Kontovas, C.A., Psaraftis, H.N. (2011), "Climate Change Policy in Shipping Focusing on Emission Standards and Technology Measures", Environmental Engineering and Management Journal, Volume 10, Issue 10, pp.1589-1596, October 2011.

REFERRED BOOK CHAPTERS

- Psaraftis, H.N., Kontovas, C.A., "Ship Emissions, Costs and their Tradeoffs", in Advances in Maritime Logistics and Supply Chain Systems, World Scientific Publishing, Singapore, 2011 Article [\[download\]](#) View/Buy at [amazon](#)
- Psaraftis, H.N, Lyridis, D.V. and Kontovas, C.A, "The Economics of Ships", in: Wayne Talley (ed): The Blackwell Companion to Maritime Economics, January 2012 View/Buy at [amazon](#)

REFERRED CONFERENCES AND SYMPOSIA

2007

- Kontovas, C.A., Psaraftis, H.N., Zachariadis, P. (2007b), "The Two C's of the Risk-Based Approach to Goal based Standards: Challenges and Caveats", International Symposium on Maritime Safety, Security and Environmental Protection, Athens, Greece, 20-21 September 2007 [\[download\]](#)
- Kontovas, C.A., Psaraftis, H.N., Zachariadis, P. (2007c), "Improving FSA as a prerequisite for risk-based GBS", PRADS 2007 – The 10th International Symposium on Practical Design of Ships and Other Floating Structures, 1-5 Oct. 2007, Houston, TX, USA [\[download\]](#)

2008

- Kontovas, C.A. and Psaraftis H.N. (2008), "Marine Environment Risk Assessment: A Survey on the Disutility Cost of Oil Spills", 2nd Int. Symposium on Ship Operations, Management and Economics, Athens, September 17-18, 2008 [\[download\]](#)
- Psaraftis, H.N., C.A. Kontovas, "CO2 Emissions Statistics for the World Commercial Fleet", Symposium of the Hellenic Institute of Marine Technology (ELINT), November 2008

2009

- Psaraftis, H.N., C.A. Kontovas, "Ship Emissions: Logistics and Other Tradeoffs", 10th Int. Marine Design Conference (IMDC), Trondheim, Norway, 26-29 May 2009 [\[download\]](#)
- Psaraftis, H.N., Kontovas, C.A., Kakalis, N., "Cost-effectiveness of Speed Reduction as an Emissions Reduction Measure for Fast Ships", 10th Int. Conference on Fast Sea Transportation, Athens, Greece, October 5 - 8, 2009 [\[download\]](#)
- Kontovas, C.A., Psaraftis, H.N., "An Online Ship Emissions Calculator as a Decision-Making Aid and Policy Evaluation Tool", IMAM 2009, International Maritime Association of the Mediterranean, Istanbul, Turkey, October 12-15, 2009 [\[download\]](#)

2010

- Kontovas, C.A., Psaraftis, H.N., "Reduction of Emissions Along the Intermodal Container Chain", 2010 Annual Conference of the International Association of Maritime Economists (IAME), Lisbon, Portugal, July 2010 [\[download\]](#)
- Kontovas, C.A., Psaraftis, H.N., "Carbon Dioxide Emissions Valuation and its Uses", The 3rd International Symposium on Ship Operations, Management & Economics, Athens, Greece 7-8 October, 2010 [\[download\]](#)

2011

- Kontovas, C.A., Psaraftis, H.N., "The Link Between Economy and Environment in the Post-Crisis Era: Lessons Learned from Slow-Steaming, European Conference on Shipping, Chios, Greece, Intermodalism & Ports (ECONSHIP2011), June 22-24, 2011 [\[download\]](#)
- Kontovas, C.A., Ventikos, N., and Psaraftis H.N. (2011), "An Updated Analysis of IOPCF Oil Spill Data: Estimation of the Disutility Cost of Tanker Oil Spills", IMAM 2011, International Maritime Association of the Mediterranean, Genoa, Italy, September 13-16, 2011
- Kontovas, C.A., Ventikos, N., and Psaraftis H.N. (2011), "Estimating the Consequence Costs of Oil Spills from Tankers, 2011 SNAME Annual Meeting & Expo, November 18-19, 2011 Houston, TX.

NON REFERRED CONFERENCES/SYMPOSIA

- Zachariadis, P., H.N. Psaraftis, C.A. Kontovas (2007a), "Risk Based Rulemaking and Design: Proceed with Caution", RINA (Royal Institution of Naval Architects, UK) conference on Developments in Classification and International Regulations, London, Jan. 2007 [\[download\]](#)

SUBMISSIONS TO THE INTERNATIONAL MARITIME ORGANIZATION (IMO) [list not exhaustive]

(Prepared, among others, by H.N. Psaraftis, C.A. Kontovas and other LMT members - Submitted by Greece)

FSA related

- "Application of the FSA Guidelines and review of FSA studies", MSC 86/17/2
- "FORMAL SAFETY ASSESSMENT: Possible Improvements on FSA Guidelines", MSC 82/INF.3, (Nov. 29-Dec. 8, 2006)
- "Formal Safety Assessment - Environmental Risk Evaluation Criteria", MEPC 56/18/1, (May, 2007)
- "An Empirical Analysis of IOPCF Oil Spill Cost Data" (Abridged Version), MEPC 60/17 - Annex 2 (July, 2009)
- "Further experience with non-linear oil spill cost functions", MEPC 61/18/2 (July, 2010)
- "Combining environmental and safety criteria and selection of a severity matrix", MEPC62/18
- "Further experience with oil spill databases and update of non-linear oil spill cost functions", MEPC62/18/1 (July 2011)

Emissions related

- "Comments on the EEDI Baseline Formula", MEPC60/4/15, (Jan. 15, 2010)