The price of Davy Jones' Locker: Assessing the social costs of marine carbon sequestration

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Disclaimer

Except where otherwise stated and acknowledged,
I certify that this Dissertation is my sole and unaided work.

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Abstract

Directly injecting CO_2 into the ocean represents one method of mitigating global climate change. However, the costs of ocean sequestration are poorly understood. The objective of this thesis is to develop a mathematical model for estimating the social costs of ocean sequestration stemming from atmospheric and oceanic damages. A model for estimating net present costs is developed as a function of depth of injection, taking the form:

$$NPC = \frac{(1-\Theta)(SCC_0)}{1-e} + \frac{\Theta}{1-e}(U+O+E) + \frac{\Theta}{1-e}\sum_0^\infty L(t)D_{t-1}(1-r(t))SCC_{t-1}b(t)$$

where:

$$b(t) = 1 + \frac{w(r(t) - r(t))}{m} + \begin{cases} 0.005(1-t/200); t < 200 \\ 0; t \geq 200 \end{cases}$$

Model simulations indicate that damages from leaked carbon dioxide potentially significant (> \$13/tC for a 2000m injection), while damages to current and potential future uses of the marine realm are inconsequential (< \$0.02/tC). Ramifications for climate policy are discussed.

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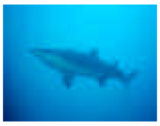
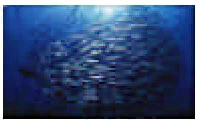
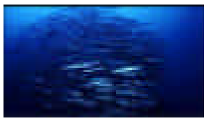
Acronyms and Abbreviations

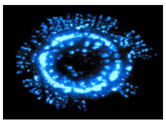
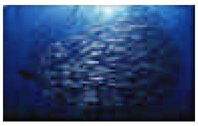
C	- Carbon
CBA	- Cost benefit analysis
CO_2	- Carbon dioxide
CV	- Contingent valuation
DEFRA	- Department for Environment Food and Rural Affairs, UK
DOE	- Department of Energy, US
FAO	- UN Food and Agricultural Organization
Gt	- Gigaton
IEA	- International Energy Agency
IGCC	- Integrated Gasification Combined Cycles
IPCC	- Intergovernmental Panel on Climate Change
mt	- Million tons
MW	- Megawatt
NOAA	- National Oceanic and Atmospheric Administration, US
NGCC	- Natural Gas Combined Cycles
NO_x	- Nitrogen oxides
NPC	- Net present costs
NPV	- Net present value
OGCM	- Ocean General Circulation Model
RCEP	- Royal Commission on Environmental Pollution, UK
SCC	- Social cost of carbon
SO_x	- Sulphur oxides
tC	- Ton carbon
WTAC	- Willingness-to-accept-compensation
WTP	- Willingness-to-pay

List of Variables

t	–	Time
NPC	–	Net present costs
L(*t*)	–	Leakage function
D(*t*)	–	Discount function
SCC(*t*)	–	Social cost of carbon function
$\mathbf{SCC_0}$	–	Social cost of carbon today
r	–	Social rate of time preference (discount rate)
?	–	Pure rate of time preference
g	–	Growth rate of per capita income
µ	–	Income elasticity of marginal utility
e	–	Base of natural logs, e = 2.71828182845904523536O3…
***b*(*t*)**	–	Growth rate of the social cost of carbon at time *t*
b	–	Marginal elasticity of environmental willingness-to-pay
t	–	Time of Leakage
G(*t*)	–	Physical damage of a unit of carbon at time *t*
b	–	Exponential connection between $pCO_2(t)$ and G(*t*)
b	–	Physical damage constant
x	–	WTP constant
e	–	Sequestration energy penalty
T	–	CO_2 capture efficiency
k	–	Electricity produced per unit carbon
N	–	Number of species in a taxon
j	–	Species richness
A	–	Area
z	–	Empirical area-species curve constant
O	–	Option value damages
U	–	Use value damages
E	–	Non-Use value damages

Chapter One – A Brief Introduction to Marine Carbon Sequestration

To me the sea is a continual miracle:
The fishes that swim—the rocks—the motion of the
waves—the ships, with men in them,
What stranger miracles are there?

-- Walt Whitman "Miracles" 1856

1.1 Background and Scope

Organic carbon buried underground as coal, oil, and natural gas is currently being combusted and released into the atmosphere at a rate of 6 gigatons of carbon per year. The ability of atmospheric carbon dioxide to absorb and reemit infrared radiation makes it an important greenhouse gas; one which is significantly altering global climate (IPCC, 2001). To address the increasing impact of climate change on human and natural systems, society presently faces a multifaceted array of mitigation and adaptation options. One such option that has recently garnered significant political and scientific attention is large-scale carbon sequestration, defined herein as the capture and storage of carbon that would otherwise be emitted to or remain in the atmosphere (Caldeira *et al.*, 2001). By storing carbon outside of the atmosphere, society can ostensibly continue to utilize fossil fuels while avoiding damaging greenhouse effects.

While the majority of research into sequestration to date has been devoted to the potential contributions of forestry and land-use change strategies, the additional storage capacity of the biosphere could only accommodate several percent of remaining fossil fuel reserves (IPCC, 2000). In contrast, the potential storage capacity of the ocean is several orders of magnitude larger; a carbon load that would double pre-industrial atmospheric concentrations would increase oceanic concentrations by less than 2% (Herzog *et al.*, 2000). As a consequence of this tremendous capacity, considerable attention is being focused on the possibility of capturing CO_2 from power plants and directly injecting it into the sea. State-sponsored research into ocean sequestration is ongoing in several nations, most notably the United States, Japan, Norway, Australia and Canada ("Project Agreement", 1997; Doyle, 2003).

Much of the current work has demonstrated that, from an engineering perspective, ocean carbon sequestration is entirely feasible (Ha-Duong and Keith, 2003). Several techniques for capturing CO_2 from industrial sources are already commercially available. These include chemical adsorption and absorption with solvents (e.g. monoethanolamine), as well as pre-combustion options such as removing nitrogen from the air prior to burning (RCEP, 2000). Once the carbon is captured, it can easily be compressed and disposed of in the ocean via ship or pipeline (Figure 1).

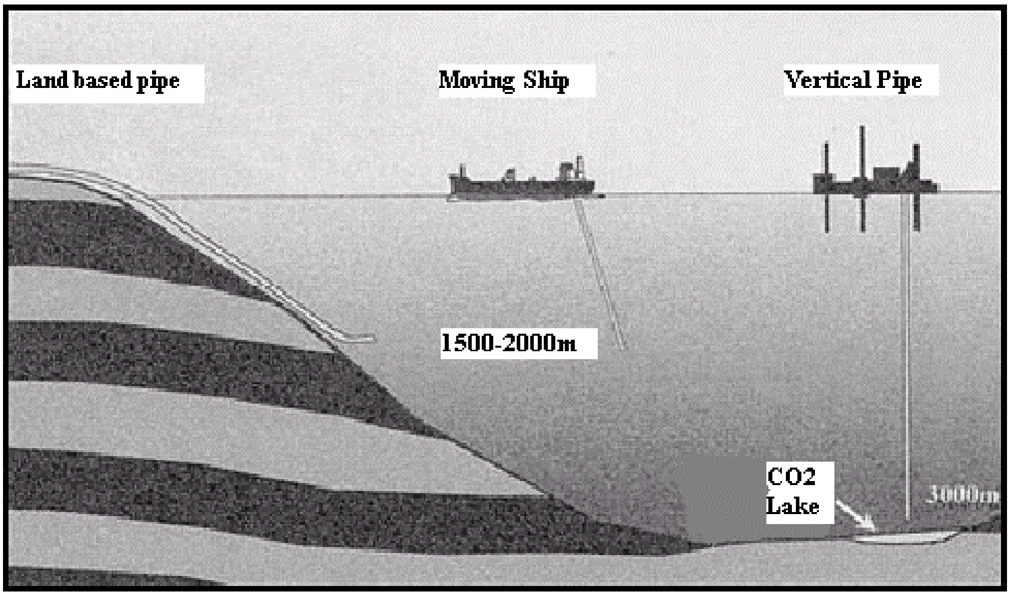


Figure 1 – Pipeline and ship options for CO_2 disposal (Johnston and Santillo, 2003).

While the engineering dimensions of sequestering carbon are relatively clear, the economic dimensions are not. Comparing the economic efficiency of ocean sequestration against other mitigation or adaptation options requires detailed analyses of the costs. Such costs include both the *private* costs associated with physically capturing and storing CO_2, and the *social* costs imposed by added or unresolved environmental externalities. Unlike private costs, social costs are particularly difficult to estimate as they are, by definition, outside the reach of Adam Smith's invisible hand. In the case of ocean carbon sequestration, social costs include both atmospheric and oceanic externalities associated with the capture and storage process. Though several reviews of the private costs of ocean carbon sequestration exist (e.g. Audus, 1997; Bock *et al.,* 2002; McFarland *et al.,* 2002; Rubin and Rao, 2002; Gielen, 2003; Davidson, 2002), comparatively little work has been directed toward developing a comprehensive estimate of social costs.

1.2 Aims and Methodology – A Roadmap

To help fill the gap in the literature, this dissertation develops an environmental and economic model for estimating the social costs of ocean sequestration. The thesis is split into two main sections, examining atmospheric and oceanic damages in turn. Chapter Two is premised on the reality that the ocean is not a permanent reservoir; carbon injected into the ocean eventually degasses into the atmosphere. A spate of recent work (e.g. Herzog *et al.*, 2003; Ha-Duong and Keith, 2003; Keller *et al.*, 2003) has attempted to evaluate future greenhouse effects of this leakage by examining the relationship between the rate at which sequestered carbon enters the atmosphere, predicted marginal damages from excess CO_2, and the discount factors applied to equate current and future costs. To date, such models have been dependent upon a number of tenuous economic assumptions, such as time-constant leakage and discount rates.

To develop a mathematical model of atmospheric damages, Chapter Two combines recent advances in economic theory on time-variable discount rates and the climatic damages, with carbon leakage rates predicted by Ocean General Circulation Model simulations. The chapter further innovates on previous models by incorporating current inefficiencies in the sequestration process which increase the social costs.

The second main thrust of the dissertation is an examination of the damages that ocean carbon sequestration is likely to have on the marine environment. Previous characterizations of the economic efficiency of ocean sequestration entirely exclude damages to aquatic ecosystems from consideration (e.g. Herzog *et al.*, 2003; Ha-Duong and Keith, 2003; Keller *et al.*, 2003). Chapter Three is a novel attempt to estimate and monetize these damages from first principles. The chapter first reviews the physical effects that injecting CO_2, a weak acid, may have on the ocean. Secondly, the analysis attempts to estimate damages from that acidification on a per ton carbon (tC) basis. To do so, valuation of the ocean is disaggregated into two distinct categories; use values and non-use values. The vulnerability of each value to carbon sequestration is then estimated. The work demonstrates that likely marginal damages to use and option values such as fisheries are inconsequential, whereas marginal damages to non-use values are potentially significant.

Findings on predicted atmospheric and oceanic damages are combined in Chapter Four – Results and Discussion. Monte Carlo simulations are employed to estimate mean values for total net present costs of ocean carbon sequestration by depth. The policy significance of these costs with respect to the economic feasibility of carbon sequestration relative to other mitigation and adaptation options is explored.

<u>Chapter Two</u> – A Leaky Locker: Developing an Economic Model for Temporary Carbon Sequestration

"The argument that we lose our souls by economically pricing the environment is silly."

-- Herendeen, 1998

2.1 Chapter Framework

This chapter is premised on the fundamental fact that marine carbon sequestration is only temporary sequestration; CO_2 intentionally injected into the ocean eventually leaks back into the atmosphere. The benefit of temporary sequestration is that it delays emissions of CO_2 into the atmosphere. As Marland *et al.* (2001) note, temporary sequestration postpones climate change damages while buying time for technological progress, capital turnover, and learning.

However, temporary sequestration bears significant costs as well. When sequestered CO_2 ultimately leaks from the ocean, it remains a greenhouse gas with all of the associated climatic effects. Estimating the costs of temporary sequestration depends, at its root, on three basic factors:

1) The rate at which carbon leaks into the atmosphere;
2) The cost of marginal damages at the time CO_2 leaks into the atmosphere;
3) The extent to which future damages are discounted against present damages;

Recent analyses have attempted to evaluate the future climatic damages from leaked carbon by examining these three factors (e.g. Herzog *et al.*, 2003; Ha-Duong and Keith, 2003; Keller *et al.*, 2003). The basic finding is that the net present costs (NPC) of sequestered carbon leaked back into the atmosphere is the product of the leakage function L(t), the discount function D(t), and the social cost of carbon function SCC(t);

$$NPC = \int_0^\infty L(t)D(t)SCC(t)dt$$

Using similar models, several authors have calculated that the economic costs of carbon leakage are potentially inconsequential. For example, Caldeira *et al.* (2001) write, "Under the assumption of a constant cost of carbon emissions and a 4% discount rate, injecting only 900m deep avoids approximately 90% of the associated global warming costs; an injection 1700m deep avoids >99% of the associated global warming costs (p. 12)." Similarly, Ha-Duong and Keith (2003) calculate that with a leakage rate of 1% and a discount rate of 4%, CO_2 sequestration avoids 80% of global warming costs.

This chapter takes issue with such optimistic findings. Work to date has been dependent on several unlikely assumptions including the application of constant leakage rates (Ha-Duong and Keith, 2003), constant discount rates (Herzog *et al.*, 2003; Keller *et al.*, 2003), constant or discount rate-independent SCC's (Caldeira *et al.*, 2001; Ha-Duong and Keith, 2003), and perfectly efficient carbon capture processes. To examine these findings, this chapter develops a model for estimating the net present costs of atmospheric damages from sequestered CO_2. The various components of future cost—L(t), D(t), and SCC(t)—are identified, with means and probability distributions estimated.

This chapter makes several advances upon previous work on the economics of ocean sequestration. First, the leakage rate section applies results from injection simulations in

Ocean General Circulation Models (OGCMs). Second, the bulk of the chapter focuses on determining the cost of leaked carbon by employing time-variable discount rates, and linking predicted discount rates to estimated growth rates in the SCC. Third, the technological inefficiencies of the carbon sequestration process are made explicit, with the energy requirements of the system and uncaptured CO_2 directly incorporated into the model.

2.2 Leakage Rates

One of the most important economic and environmental considerations with respect to sequestration is the speed at which disposed carbon leaks back into the atmosphere. Research into the issue of the leakage rate can be descriptive or normative, focusing on either actual or optimal leakage rates in light of uncertainty.

Most of the work on leakage and sequestration to date has not focused on ocean sequestration *per se*. Instead, the majority of analyses of temporary storage have developed normative leakage rates, $L(t)$, based on an exponential decay model. For example, Keller *et al.*'s (2003) economic description of sequestration starts with a basic model wherein leakage is a function of the decay rate (Q), such that;

$$L(t) = \frac{Q}{e^{Qt}}$$

Exponential decay rates, while perhaps applicable to storage in the lithosphere or biosphere, do not accurately reflect predicted patterns of oceanic leakage. An exponential decay model has peak leakage occurring at time zero, and asymptotically declining thereafter. However, carbon injected into the ocean at depth must slowly diffuse through the deeper layers of the water column. As a result, there is a considerable time-lag before carbon reaches the ocean surface, and peak leakage rates may not occur for decades or centuries after the time of injection. While leakage rates depend on a number of site-specific variables such as sub-surface currents and the availability of bicarbonate, in general, the deeper carbon that is injected, the slower the overall rate.

The current analysis employs a descriptive leakage model developed by Caldeira *et al.* (1998; 2001). Their study estimates leakage rates of injected carbon based on both a one-dimensional box-diffusion model and a three-dimensional OGCM, using the OGCM model to help validate the simpler model. The authors simulate CO_2 injections at several locations, assuming background atmospheric CO_2 concentrations described in the IPCC S650 scenario (stabilization at 650 ppm in 2200). The results of Caldeira *et al.*'s work are presented in Figure 2.

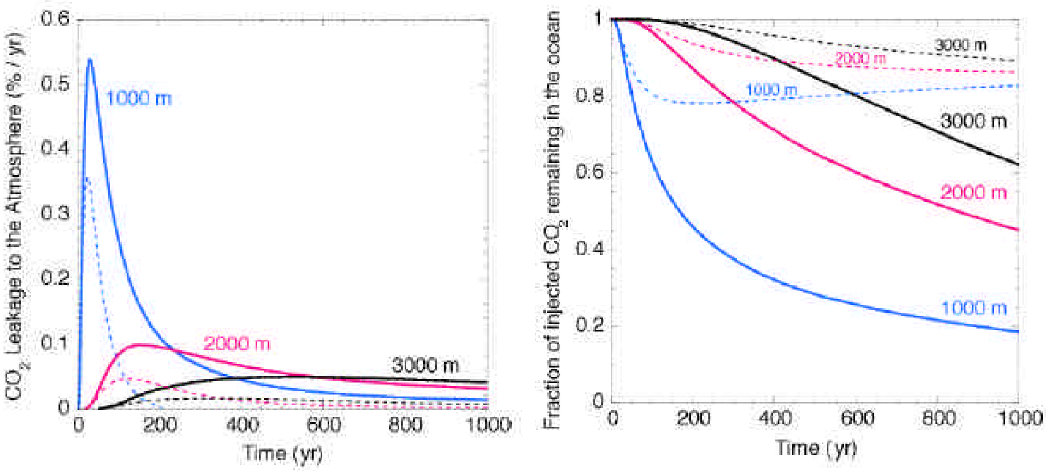


Figure 2 – Predicted leakage rates from ocean sequestration by disposal depth (Caldeira *et al.*, 2001). Dotted lines represent net carbon fluxes; e.g. they include the *re-absorption* of leaked carbon by the ocean.

When carbon is injected at 2000m or greater, the annual flux of carbon leaked back to the atmosphere never exceeds 0.1% of the initial injection, and only peaks after 100 to 150 years. One important caveat with respect to leakage concerns the equilibrium between the ocean and atmosphere. Because the two systems are in constant contact and are driven by chemical and physical parameters, roughly 80% of the carbon that degasses into the atmosphere will ultimately be reabsorbed by the ocean (Herzog *et al.*, 2003). As reabsorption is a natural process the economic efficiency models are only concerned with the initial leakage.[1]

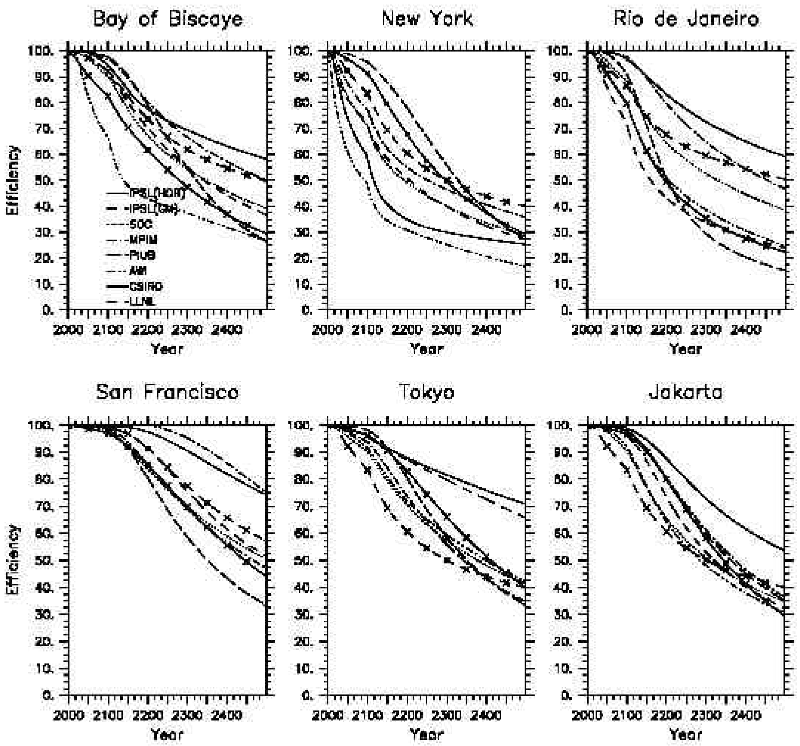


Figure 3 – Estimated retention efficiencies for a 1500m injection over 500 years, as calculated with 8 OGCM models at six locations (Orr *et al.*, 2002).

[1] A secondary effect of ocean injection is a potential dampening effect have on the ocean's ability to remove carbon from the atmosphere. Sequestering carbon in the ocean depletes carbonate ions, reducing the effectiveness of oceanic uptake, thereby increasing the half-life of atmospheric carbon (Bacastow *et al.*, 1997). This effect drives up the social cost of carbon. The modeling in this exercise is complicated, and represents a potentially important area of future research.

Results from other OGCMs share leakage patterns similar to Caldeira *et al.*'s, but yield slightly different rates (Mignone *et al.*, 2002). For example, Orr *et al.* (2000) simulated ocean injection in eight different OGCMs. They found that, for carbon injected at 1500m, estimated retention efficiency was 82-96% after 100 years and 28-57% after 500 years, due to both inter-model and inter-site variation (Figure 3). The inter-model variation is potentially due to subgrid-scale processes, particularly differing parameterizations of vertical exchange within the low-latitude pycnocline (Mignone *et al.*, 2002). To incorporate some of the parameterization uncertainty into the economic analysis, this model will employ Caldeira *et al.*'s estimates of leakage rates as a central figure, but apply a standard deviation of +/- 10% of the mean (Table 1).

Table 1 – Predicted carbon leakage rates based on a visual interpolation of Figure 2.

Depth	**Period (yrs)**	**Mean Leakage rate (%, t = yr)**
1000 m	0 – 25	(.0216) t
	26 – 100	0.54 – 0.0032 (t – 25)
	101 – 200	0.30 – 0.002 (t – 100)
	201 – 400	0.1 – 0.00025 (t – 200)
	401 – 1000	0.05 – 0.00005 (t – 400)
2000 m	0 – 25	0
	26 – 150	0.0008t – 0.02
	151 – 1000	0.1 – 0.0000824 (t – 150)
3000 m	0 – 50	0
	51 – 400	0.000114 (t – 50)
	401 – 1000	0.04

2.3 The Discount Factor and the Social Cost of Carbon

While estimating leakages rate is an exercise in modeling a geochemical system, determining what value to place on that leakage is an economic analysis carrying considerably greater uncertainty. At its heart, the valuation has two basic components: the estimated social cost of carbon at the time at which it degasses into the atmosphere, SCC(t), multiplied by a time-variable discount factor used to equate future costs with current costs, D(t). The rate of change in the discount factor over time is known as the discount rate. Previous analyses (e.g. Herzog *et al*., 2003; Keller *et al.*, 2003; Ha-Duong and Keith, 2003) have treated the discount rate and cost of carbon as independent variables. This analysis will examine each separately, as well as likely correlations between the two.

2.3.1 The Discount Factor and Discount Rate

The discount factor at time t, D(t), reflects the weighting placed on costs or benefits that occur at time t relative to current prices. The discount rate, r, is the rate at which the discount factor decreases over time. For the purpose of this discussion, the "discount rate" is synonymous with the "social rate of time preference" which can be disaggregated into two parts: the pure rate of social time preference (consisting of impatience and the potential for social catastrophe), ?, plus the growth rate of per capita income, g, multiplied by the income elasticity of marginal utility, µ (UK Treasury, 2002; Clarkson and Deyes, 2002):

$$r = ? + \mu g$$

The significance of ?, µ, and g will be further discussed in the follow section on the SCC. For the time being, discussion will focus on the discount rate, r, in aggregate.

Because oceanic leakage occurs over a time-scale of hundreds of years, damage estimates are extremely sensitive to the compounding effect of the discount rate. If the discount rate is zero and the SCC is constant, there is no economic advantage to temporary sequestration; all of the carbon eventually returns to the atmosphere at the same net present value. However, as the discount rate increases, future damages are devalued.

Take for example a hypothetical reservoir that stores carbon perfectly for a century then releases it all into the atmosphere. Assuming a constant SCC (say \$100/tC) and a discount rate of zero, there is no advantage to delayed over immediate emissions. Both cost \$100/tC. However, applying a 1% constant discount rate reduces the net present costs of the future release to \$37/tC. A 5% rate would further contract them to 50¢. As oceanic leakage occurs over centuries, the compounding effect of the discount rate plays a critical role in determining the economic effectiveness of sequestration.

Despite its significance, there is little agreement on what discount rate is appropriate to use. Most analyses of carbon disposal have treated the discount rate as a key independent variable and provided a range of results, rather than attempting to predetermine the appropriate rate (e.g. Keller *et al.*, 2003; Caldeira *et al.*, 2001; Herzog *et al.*, 2003). For example, Caldeira *et al.* (2001) and Herzog *et al.* (2003) plot the economic potential of ocean disposal as a function of the discount rate and depth, though they select 3% as the base case. Keller *et al.* (2003) also recognize the uncertainty in discount rates, but set a best guess around a discount rate of 5%. Others (e.g. Marland *et al.*, 2001) have adopted a discount rate of zero. A common thread is that all of the sequestration analyses to date have been dependent upon constant discount rates, while providing little guidance on an *appropriate* rate to use.

This analysis improves upon past sequestration work by attempting to determine an appropriate discount rate under conditions of uncertainty through hyperbolic discounting. This effort is informed to a great extent by the remarkable efforts of Weitzman (2001). The Harvard economist solicited the "professional considered gut feeling" about an appropriate discount rate to use for climate change from 2,100 economists worldwide. Responses ranged from –3% to 27%(!), forming a gamma distribution with a mean of 4% and a standard deviation of 3%. Given this distribution, Weitzman determined a certainty-equivalent rate by calculating back from the mean discount factor over time. The certainty-equivalent discount rate starts at the mean rate (4%), and gradually falls to the lowest estimates (0%), as the higher rates discount away their own relevance over time. There are at least three rationales for such hyperbolic discounting (see, e.g., Weitzman, 2001; Groom *et al.*, 2003; Heal, 1997).

1) The discount rate is predicated on futures rate of economic growth. Because future growth rates are uncertain, a range of discount rates is possible, with certainty-equivalent rates approaching the lower boundary over time. Newell and

Pizer (2003) support Weitzman's analysis with empirical evidence from U.S. interest rates. Similar to Weitzman's findings, their analysis suggests that if the growth rate follows a random walk path, the certainty-equivalent rate falls from 4% to 2% after 100 years, and to 0.5% after 300 years (Figure 4).

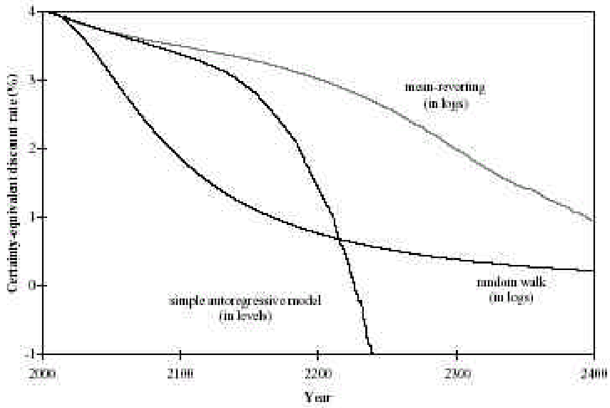


Figure 4 – Forecasts of certainty equivalent discount rates derived from U.S. treasury bonds. Rates in the random walk model approximate Weitzman's suggested rates (Newell and Pizer, 2003).

2) There is an ethical justification, based on the issue of intergenerational equity (Pearce *et al.*, 2003). Constant discount rates have the effect of minimizing the significance of future generations in the current calculus. As Weitzman wrote, "To think about the distant future in terms of standard discounting is to have an uneasy intuitive feeling that something is wrong, somewhere (1998; p. 201)."
3) Individuals tend to use declining discount rates on a personal basis, which strongly supports hyperbolic discounting (Frederick *et al.*, 2002). Pearce *et al.* (2003) argue that if individual preferences reflect hyperbolic discounting, social preference can legitimately do the same.

The application of declining discount rates is important because it significantly increases the valuation of future carbon leakage. For the purpose of this analysis, a time-variable discount rate will be applied, in keeping with recent UK Treasury policy (UK Treasury, 2002). As a mean best estimate, the certainty equivalent rates of professional opinion suggested by Weitzman (2001) are applied. The two boundary conditions selected here are; 1) a lower boundary of a constant discount rate of zero, and; 2) an upper boundary of time-variable rates double those suggested by Weitzman.

Table 2 – Modeled discount rate schedule

Period (yrs)	Lower Boundary (%)	Mean (%)	Upper Boundary (%)
0 – 5	0	4	8
6 – 25	0	3	6
26 – 75	0	2	4
76 – 300	0	1	2
300 – 1000	0	0	0

2.3.2 The Social Cost of Carbon (SCC) and SCC Growth Rate, *b*

The third factor in estimating the cost of emissions at time t is the social cost of carbon SCC(t). This term represents the cost of the aggregated damages caused by a unit of carbon emitted into the atmosphere, stemming from the effects that climate change is likely to have on the economy, human health, and the environment (IPCC, 2001). The SCC is not constant, but varies in time as the composition of the atmosphere and the economy change.

There are two key variables that need to be teased apart for a discussion of the SCC; the SCC at present, SCC_0, and the rate at which the SCC is predicted to grow in the future, b. If the growth rate b is constant, then:

$$SCC(t) = SCC_0(e^{bt})$$

For our discrete-time model, this can be approximated as:[2]

$$SCC(t) = SCC_0(1+b)^t$$

2.3.2.1 *Estimating SCC_0*

Selecting an appropriate initial value for the SCC is a hotly contested issue, with widely ranging estimates. The Intergovernmental Panel on Climate Change (IPCC, 1996) notes that estimates of the SCC range between \$9/tC and \$190/tC for 2001-2010, a factor of more than twenty. Many of these studies are presented in Table 3. Two recent reviews help to explain this tremendous variation. The first, commissioned by the UK Department for Environment Food and Rural Affairs (DEFRA) surveys the multiplicative layers of uncertainty in calculating a value for the SCC, including uncertainty in future emissions, uncertain impacts of CO_2 on climate, the use of the marginal cost method or the cost-benefit approach method, uncertain effects of climate on the economy, and the unknown potential for catastrophe, adaptation, and socially contingent damages (Clarkson and Deyes, 2002). These uncertainties are further complicated by two equity issues that significantly affect the valuation of damages: inter-temporal equity (reflected in the selection of an appropriate discount rate), and spatial equity (reflected in the selection of an appropriate equity weighting). The DEFRA authors suggest applying a mean or "defensible illustrative" value of £70/tC (\$110/tC), within a sensitivity range of £35 –

[2] For the reason that $\lim_{m\to\infty}(1+bm)^{mt} = e^{bt}$.

£140/tC (\$56 - \$224/tC), for emissions in 2000 (Clarkson and Deyes, 2002).

The second work, a critical review by Pearce (2003a) takes issue with the DEFRA assessment, advocating instead a lesser range of £4 - £27/tC (\$6 - \$41/tC) that incorporates an equity weighting and time-variable discount rates. The main difference between the DEFRA paper and Pearce's is the selection of a baseline economic model. DEFRA's estimates are based on the work of Eyre *et al.* (1999) which ignores the potential for economic adaptation to climate change (the "dumb farmer" effect). Pearce, in turn, prefers a model by Tol and Downing (2000) which includes adaptation.

As adaptation is undoubtedly an important aspect of damage estimates, this analysis will employ Pearce's more conservative estimate of \$6 - \$41/tC for SCC_0, with a gamma distribution over the central estimate of \$23/tC. However, the \$6 - \$41/tC range needs to be adjusted upwards as Tol and Downing's model excludes the potential for "large scale disruptions, such as a breakdown of North Atlantic Deep Water formation or a collapse of the West-Antarctic Ice Sheet (Tol and Downing, 2000; 22)." Such catastrophic damages appear to be extremely important for damage estimates. For example, Gjerde *et al.* (1999) finds that the emissions reductions required for potential catastrophic damages exceed the corresponding reductions from continuous damages. A more recent analysis by Pizer (2003) estimates an optimal carbon tax of \$8/tC in 2010 for continuous damages, but a tax of \$35/tC if catastrophic damages are considered; a factor of four. Moreover, Pizer's optimal tax levels rise to \$30/tC and \$500/tC respectively in 2060; a factor of 17! Given that potential catastrophic damages appear to be at least as important as continuous damages, one can conservatively adjust Pearce's suggested SCC_0 by doubling the expected damage range to $12-$82/tC, with a central estimate at $47/tC.

Table 3 – Estimates of the SCC by period, and derived annualized growth rates (data from Pearce, 2003a, and Clarkson and Deyes, 2002)

Author, year; discount rate; method	**SCC ($/tC); 1991-2000**	**Annual growth rate**	**SCC ($/tC); 2001-2010**	**Annual growth rate**	**SCC ($/tC); 2011-2020**	**Annual growth rate**	**SCC ($/tC); 2021-2030**	**Overall growth rate**
Nordhaus, 1994; ? = 3, best guess; CBA	7.2	2.48	9.2	2.345	11.6	1.53	13.5	**2.1**
Nordhaus, 1994; ? = 3, expected value; CBA	16.2	4.14	24.3	0	24.3			**2**
Nordhaus and Boyer, 2000; *r* = 3; CBA	6.4	3.58	9.1	2.72	11.9	2.34	15	**2.9**
Fankhauser, 1995; ? = 0.5; MC	27.4	1.18	30.8	1.05	34.2	0.93	37.5	**1.1**
Fankhauser, 1995; ? = 0; MC	65.6						84.5	**0.85**
Fankhauser, 1995; ? = 3; MC	7.3						11.1	**1.4**
Cline, 1993; *r* = 0; CBA	167.5	2.19	208	1.9	251.2	1.74	298.5	**1**
Cline, 1993; *r* = 10; CBA	7.8	2.82	10.3	2.51	13.2	1.88	15.9	**2.4**
Peck/Teisberg, 1992; ? = 3; CBA	13.5-16.2	1.84	16.2-18.9	1.553	18.9-24.3	2.54	24.3-29.7	**2**
Maddison, 1994; ? = 5; CBA	8	3.14	10.9	3.24	15	2.87	19.9	**3.1**
Maddison, 1994; ? = 5; MC	8.2	3.23	11.3	3.21	15.5	2.84	20.5	**3.1**
Tol, 1999; *r* = 5; MC	14.9	1.62	17.5	1.445	20.2	1.87	24.3	**1.6**

Eyre et al, 1999. r = 1; no equity, OF; MC	110 (1995-2004)	0.88	120 (2005-2014)						**0.88**
Eyre et al, 1999. r = 3; no equity, OF; MC	53 (1995-2004)	1.74	63 (2005-2014)						**1.7**
Eyre et al., 1999. r = 5; no equity, OF; MC	37 (1995-2004	2.42	47 (2005-2014)						**2.4**
Tol, 1999. r = 1; no equity, FUND 1.6; MC	109 (1995-2004)	0.88	119 (2005-2014)						**0.88**
Tol, 1999. r = 3; no equity, FUND 1.6; MC	42 (1995-2004)	1.55	49 (2005-2014)						**1.6**
Tol, 1999. r = 5; no equity, FUND 1.6; MC	20 (1995-2004)	2.26	25 (2005-2014)						**2.3**
Roughgarden and Schneider, 1999; lower bound; CBA	6.7	1.92	8.1	2.91	10.8	2.23	13.5	**2.4**	
Roughgarden and Schneider, 1999; upper bound; MC	14.9	1.62	17.5	2.13	21.6	2.77	28.4	**2.2**	
Clarkson and Deyes, 2002; r = 3; MC.	112	1.23	128	1.12	144	1.01	160	**1.1**	

2.3.2.2 *Estimating b*(t)

While SCC_0 is clearly important for the economics of sequestration, when assessing the economic benefits of temporary sequestration relative to direct emissions SCC_0 may be less important than the *rate of change* of the SCC. Since disposed carbon ultimately leaks back into the atmosphere, the rate of growth in the valuation of future emissions is critical. In general the SCC is predicted to rise, for at least two reasons. First, CO_2 is a stock pollutant with increasing marginal damages (Nordhaus, 1994). Leakage occurs in the future when atmospheric concentrations will be higher, and therefore marginal damages will be higher. Second, as income grows, social willingness-to-pay to avoid environmental damages is likely to increase as well; the theory of environmental Kuznets curves dictates that richer societies tend to be increasingly willing to spend more to reduce pollution.

In keeping with this prediction, current economic models of the SCC estimate positive values for *b*. This analysis indicates that the dozen studies (21 scenarios) surveyed between Pearce (2003a) and Clarkson and Deyes (2002) predict average annualized *b* values between 0.85% and 3.1% over 30 years (Figure 5), despite typically excluding the potential for catastrophic damages. Though each study reveals some decadal variation in *b*, there is no clear trend of either increasing or decreasing rates.

Given its depends on atmospheric CO_2 concentrations and economic growth, *b* itself is logically time-variable. However, there is clearly a difficulty in estimating an appropriate growth rate for *b*, when CO_2 leakage is likely to peak *centuries* after the time-horizons of economic models. Moreover, the analysis of *b* is complicated by the fact that large-scale sequestration strategies will change the evolution of the SCC. Non-marginal sequestration strategies can reduce short- and mid-term *b* values by decreasing atmospheric CO_2 concentrations relative to the baseline scenario. However, if sequestration is employed *instead* of efficiency and renewable energy sources, short- and mid-term damages remain comparable while the long-term damage curve is driven up due to oceanic leakage. For example, Ha-Duong and Keith (2003) calculate that, "At the global scale, if industrial carbon management plays a big role in mitigating emissions, then as much as 500 GtC could be stored by 2100. If the average leak rate is only 0.2% annually, there would be a 1 GtC per year source undermining CO_2 stabilization (p. 8)."

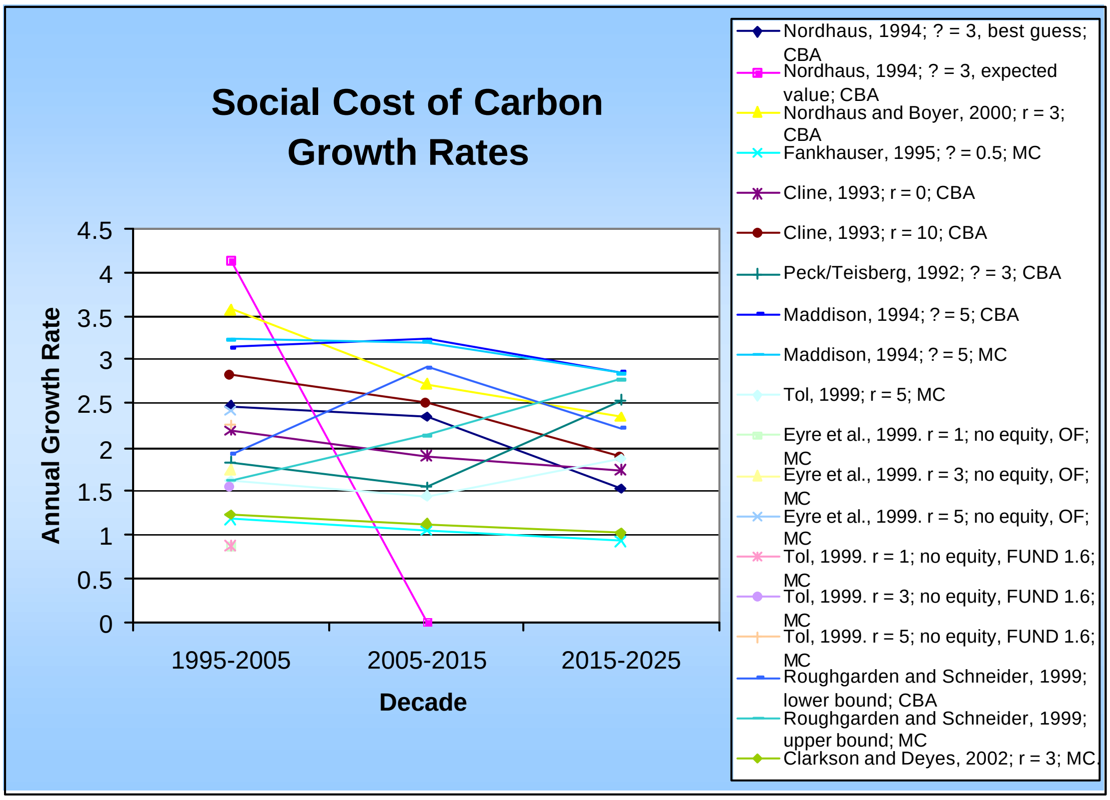


Figure 5 – Predicted growth in the social cost of carbon by decade (calculated from data summarized by Pearce, 2003a and Clarkson and Deyes, 2002)

One means of dealing with this uncertainty would be to use certainty-equivalent rates of growth of the SCC, as was done with the discount rate. Such an approach would start with a range of estimates for *b*, and determine certainty-equivalent future growth rates through average discount factors. In contrast to discount rates, the certainty-equivalent rate for b would approach the higher boundary, due to the compounding effect of $(1+b)^t$, where $b > 0$. Unfortunately, this strategy is less applicable to estimating the rate of growth of the SCC due to several factors including the relative scarcity of estimates (a dozen compared to 2,160 estimates) and the time-dependent nature of b.

Therefore, an alternative approach for estimating b is developed here. The basic premise is that the discount rate, $r(t)$, and growth in the SCC, $b(t)$, are both dependent on assumptions about economic growth, g. The predicted certainty-equivalent discount rates, r, can therefore be used as a proxy for measuring b.

2.3.3 Connections between the discount rate, $r(t)$, and SCC growth, $b(t)$

The connection between discount rates and SCC_0 is an obvious one, which has significantly raised the profile of discount rates in the climate change mitigation debate (e.g. Groom *et al.*, 2003). Indeed, the choice of a discount rate is perhaps the single most important variable in determining SCC_0 (Clarkson and Deyes, 2002), as higher rates reduce the magnitude of SCC_0 by discounting away future impacts. However, the connection between the discount rate and *the rate of growth* of the SCC, b, is an unresolved issue in environmental economics; it represents a significant theoretical problem.

To date, studies on the economic value of temporary sequestration have been largely silent on the relationship between b and r. One exception is the work of Herzog *et al.* (2003) which correctly holds that the critical factor in determining the economic potential of a leaky reservoir is the ratio between the two variables. Herzog *et al.* outline three "limiting carbon price assumptions" for a given discount rate:

1) SCC(*t*) remains constant, due to constant marginal damages. With constant costs, the value of sequestration is potentially very large due to the compounding effect of the discount rate.
2) SCC(*t*) rises at the same rate as the discount rate, based on a Hotelling model in which carbon storage in the atmosphere represents a limited natural resource. If the cost of carbon rises at the discount rate, temporary sequestration holds little economic value.
3) SCC(*t*) rises with the discount rate until a time, *t**, when a backstop technology becomes competitive (e.g. removal of carbon directly from the atmosphere). The backstop technology causes the cost of carbon to remain constant thereafter or fall.

However, Herzog *et al.*'s assumption that the rate of change of the SCC is necessarily bounded by zero and the discount rate is erroneous. The Hotelling rule applies to efficient non-renewable resource allocation under perfect market or monopoly conditions; however, it's applicability to sequestration is limited. Reductions in extraction costs have caused real prices of most non-renewable resources to fall significantly over the past century (e.g. petroleum, coal, copper, lead, aluminium, sulfur, phosphorus, and even farmland), indicating that technological progress could conceivably cause b to be negative over time (Nordhaus, 1992). Alternatively, Boiteux (1976) argues that the rate at which the prices of natural resources grow can *exceed* the discount rate as a result of changing social preferences, such as a growing willingness-to-pay (Philibert, 2003).

Moreover, the applicability of the Hotelling rule is limited by imperfect information. Climatic unknowns (natural variability, feedback effects, thresholds, etc.) limit our understanding of the shape of the marginal damage curve (IPCC, 2001). Learning can cause dramatic swings in the SCC; for example, *b* would increase dramatically and non-linearly were a catastrophic threshold discovered. In support of this notion, Pizer (2003) suggests that the potential for catastrophe creates a growth rate in optimal carbon prices of 5.5% between 2010 and 2060.

To help fill this gap in existing economic theory, a model of *b* is developed in a side paper (Elliott and Hepburn, 2003; *mimeo*). To start, SCC at time t is the aggregated damage curve of a unit of carbon emitted/leaked into the atmosphere at t. Damages are the sum of annual costs of carbon damages, C(*t*), discounted to net present value in the year of emission through the function, *D*(*t*-*t*);

$$SCC(t) = \int_{t}^{\infty} C(t) D(t - t) dt$$

Costs of damages in a given year can be disaggregated into two parts, the physical damage function, G(*t*), multiplied by society's willingness-to-pay to avoid those damages, WTP(*t*).

$$C(t) = \Gamma(t) WTP(t)$$

Hence:

$$SCC(t) = \int_{t}^{\infty} \Gamma(t) WTP(t) D(t - t) dt$$

In turn, WTP can be described as a function of growth in per capita income, g, a constant, x, and the income elasticity of willingness-to-pay, w (Pearce *et al.*, 2003);

$$WTP(t) = x \cdot e^{(wg)(t)}$$

The discount function, D (t-t), as defined in the previous section, is equal to $e^{-r(t-t)}$. Hence;

$$SCC(t) = \int_{t}^{\infty} \Gamma(t) \cdot xe^{[wg(t)]-[r(t-t)]} dt$$

And total net present costs equal:

$$NPC = \int_{0}^{\infty} L(t) D(t) \int_{t}^{\infty} \Gamma(t) \cdot xe^{[wg(t)]-[r(t-t)]} dt dt$$

Substituting (? + µg) for r, yields:

$$NPC = \int_{0}^{\infty} L(t) D(t) \int_{t}^{\infty} \Gamma(t) \cdot xe^{[wg(t)]-[(r+mg)(t-t)]} dt dt$$

Growth in the SCC will therefore occur if the damage function, G(t), increases with time (dG(t)/dt is positive), or as the time of leakage increases due to the compounding effect of WTP, $e^{wg(t)}$ (assuming that elasticities are time-constant). Changes in the rate of economic growth, g, will affect the SCC differentially, depending on the size of t and the relationship between w, ?, and µ. The damage function and this relationship are further examined below.

2.3.3.1 *Estimating G(t)*

The aggregate physical damages from a unit of carbon leaked into the atmosphere at time t are a function of two fundamental components. The first component is the residence time of CO_2 in the atmosphere—longer residence times increase aggregate damages. A reasonable estimate of the current half-life of atmospheric CO_2 is 19-49 years, with a best estimate of 31 years (Moore and Braswell, 1994). Assuming a 31 year half-life, the remaining fraction in the atmosphere at time t is equal to $e^{-0.0224(t-t)}$.

The second component is the marginal damage caused by any unit of carbon in the atmosphere at a given time. Clearly, physical damages are linked to the excess atmospheric concentration of CO_2, but the nature of that linkage is unclear (IPCC, 2001; Clarkson and Deyes, 2002). For example, damages may increase linearly relative to atmospheric CO_2 concentrations, such that:

$$\Gamma(t) = e^{-0.0224(t-t)} d[CO_2(t)]$$

Or, damages may increase exponentially (raised to the exponent ?), such that;

$$\Gamma(t) = e^{-0.0224(t-t)} d[CO_2(t)]^{g}$$

This model will conservatively assume that damages increase linearly (? = 1), but also examine the effect of damages increasing parabolically (? = 2) in the analysis section. For consistency purposes with the leakage models, IPCC S650 is applied as the reference case (Figure 6).

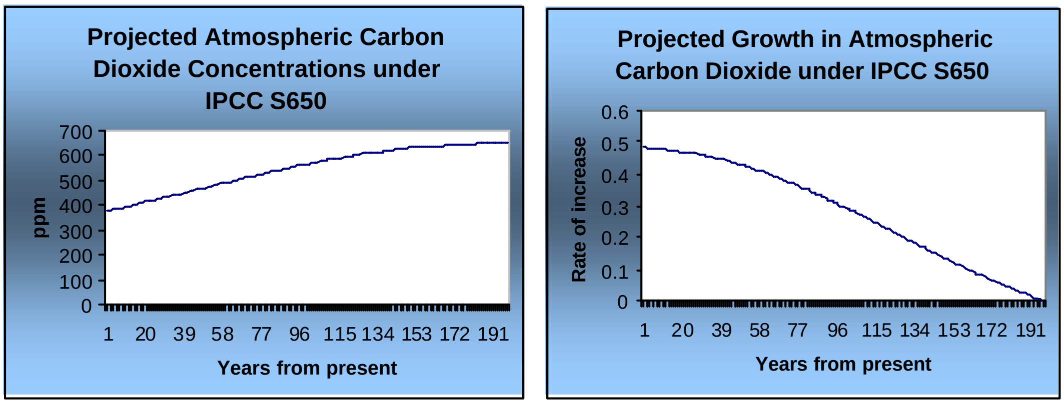


Figure 6 – Atmospheric CO2 concentrations and growth rates under IPCC S650 (data – IPCC, 2001).

Concentrations of CO_2 in IPCC S650 grow exponentially but at a linearly decreasing growth rate, which starts at 0.5% and reaches 0% 200 years from now. After year 200, CO_2 concentrations are constant at 650 ppm, or 1.73 times present concentrations. Hence, for 0<t<200, the following discrete time function applies;

$$CO_2(t) = [CO_{2(0)}][e^{a(t)}]$$

If a'(t) reflects the linearly declining growth rate in carbon concentrations described above, and $e^{0.55} = 1.73$, then:

$$a'(t) = \max\left[\frac{5}{1000}\left(1 - \frac{t}{200}\right), 0\right]$$

And:

$$a(t) = \begin{cases} \frac{5.5t}{1000}\left(1 - \frac{t}{400}\right); t < 200 \\ 0.55; t \geq 200 \end{cases}$$

This substitutes into the damage formula as:

$$\Gamma(t) = e^{-0.0224(t-t)} d[CO_{2(0)}][e^{\frac{5.5t}{1000}(1-\frac{t}{400})}] \text{; for } t < 200;$$

$$\Gamma(t) = e^{-0.0224(t-t)} d[CO_{2(0)}](1.73) \text{; for t} \geq 200.$$

In summary, the current expression for net present costs is the unwieldy formula:

$$NPC = \int_0^\infty L(t)D(t)\int_t^\infty x e^{[wg(t)]-[(r(t)+mg(t)+0.0224)(t-t)]} \begin{cases} d[CO_{2(0)}]e^{\frac{5.5t}{1000}(1-\frac{t}{400})}; t < 200 \\ d(1.73)[CO_{2(0)}]; t \geq 200 \end{cases} dtdt$$

Given the intractable nature of estimating the constants d and x, one can approximate the equation by assuming that SCC_0 will grow at the rate of atmospheric CO_2, $d[CO_2]/dt$, plus the rate of growth in WTP, *wg.* This effectively disregards effects that changes in certainty-equivalent future discount rates are likely to have on the shape of the aggregate damage curve. However, this is mitigated to some extent in that declining rates have already been incorporated into the estimate of SCC_0. Moreover, this assumption becomes increasingly justifiable for leakage in the distant future, when $t >> (t - \mathrm{t})$. Thus, the approximated version of $b(\mathrm{t})$, is;

$$\begin{cases} b(t) = 1 + wg + 0.005(1 - t/200); t < 200 \\ b(t) = 1 + wg; t \geq 200 \end{cases}$$

Hence, the key factor in determining the long-term evolution of b and hence SCC(t) is the willingness-to-pay growth rate, wg. The following section attempts to estimate values for w and g. As g is a major component of both WTP growth (wg) and discount rates (r = ? + µg), assumptions about g need to be consistent with previous assumptions made when estimating r.

2.3.3.2 *Estimating w*

Estimates of the marginal elasticity of environmental willingness-to-pay, *w*, are a significant uncertainty, as there are few available studies of income elasticity for environmental goods. Reported income elasticities of willingness-to-pay for environmental goods generally fall between zero and one (Flores and Carson, 1997; Horowitz and McConnell, 2002). For example, the IPCC documents household elasticities in the 0.2-0.6 range based on income differentials *within* a country (IPCC, 2001), while Kristrom and Riera (1996) survey elasticities of 0.2-0.3 for environmental goods in Europe.

However, in contrast to these relatively low estimates, estimates of *social* income elasticities of WTP appear significantly higher (Pearce, 2003b). This finding is reflected in the observation that expenditure on environmental protection typically rises at a greater rate than growth in national GDP; e.g. Pearce and Palmer (2001) find an elasticity of 1.2 for environmental expenditures for EU countries. Moreover, environmental goods are often considered to be luxury goods, implying elasticities greater than one. Pearce (2003b) concludes that income elasticity of WTP for environmental change is less than one, and "numbers like 0.3-0.7 seem about right (p. 35)." For this model, a normal distribution around a mean *w* of 0.5, the central figure in Pearce's range, will be employed, but with a wider range of 0.2-1.2.

2.3.3.3 Estimating g(t)

To estimate *g* one could conceivably rely on empirical rates of growth in personal income for guidance. For example, such rates averaged approximately 2% in the UK over recent decades (Clarkson and Deyes, 2002), and Tol and Downing (2002) likewise assume an average world economic growth rate of 2.2% through 2100. However, the notion that such rates will continue in perpetuity depends on unlikely assumptions about future returns on capital; as Newell and Pizer (2003) note, growth rates are uncertain over time, particularly in the distant future. To be consistent with the previous discussion of discount rates, one can express $g(t)$ in terms of the estimated discount rate, $r(t)$.

Rearranging:

$$r(t) = r(t) + mg(t)$$

One finds that:

$$g(t) = \frac{(r(t) - r(t))}{m}$$

Thus;

$$wg(t) = \left(\frac{w}{m}\right)(r(t) - r(t))$$

To incorporate this formula into the damages model, estimates for each of these variables are made below.

2.3.3.4 *Estimating ?(t)*

Selecting a value for the pure rate of time preference, ?, is a complicated issue, with the fundamental debate over the application of ? for social decisions ranging back decades.

For example, Frank Ramsey, one of the founders of dynamic economics, argued in 1928 that discounting with ? “is ethically indefensible and arises merely from the weakness of imagination,” while Roy Harrod, his contemporary, characterized the practice as a “polite expression for rapacity and the conquest of reason by passion” (Ramsey, 1928; Harrod, 1948; quoted in, Heal, 1997). In contrast, most modern economists apply a value of ? in dynamic analyses. The UK Treasury, for example, suggests employing an initial ? of 1.5% for social decisions (2002). Given this uncertainty in ?, the certainty-equivalent ?, like r, declines over time (Li and Lofgren, 2000). As ? < r (assuming positive growth), values for ?(t) can be approximated as a preset fraction of $r(t)$, such that ?(t) declines as $r(t)$ declines. For this model, a central estimate of ?(t) will be set at $0.25r(t)$, or currently 1%, with a standard deviation of $0.125r(t)$ and boundaries of zero and $r(t)$.

2.3.3.5 *Estimating µ*

The income elasticity of marginal utility, µ, is also a matter of debate, as utility is inherently difficult to measure. Both Clarkson and Deyes (2002) and Pearce *et al.* (2003) note the discussion over the value of µ, but indicate that recent reviews suggest a value of around 1.0. For a broader range, this analysis will employ Pearce’s (2003a) advice that, “values in the range 0.5 to 1.2 seem reasonable (p. 16)”.

Applying these mean estimates, one finds that the current predicted growth rate of the SCC in this model is equal to:

$$d[CO_2]/dt + \left(\frac{w}{m}\right)(r(t) - r(t) = 0.5\% + \left(\frac{0.5}{1.0}\right)(4\% - 1\%) = 2\%$$

This estimate fits neatly with the range of *b*'s predicted by the SCC models listed in Table 3. One also finds empirical support for this formulation in a positive correlation between *b* and *r,* since *g* partially underpins both. The SCC models sampled by Pearce (2003a) and Clarkson and Deyes (2002) show a general upward trend in *b* as sampled discount rates are increased (Figure 7). This trend is even clearer when one looks within studies (Figure 8).

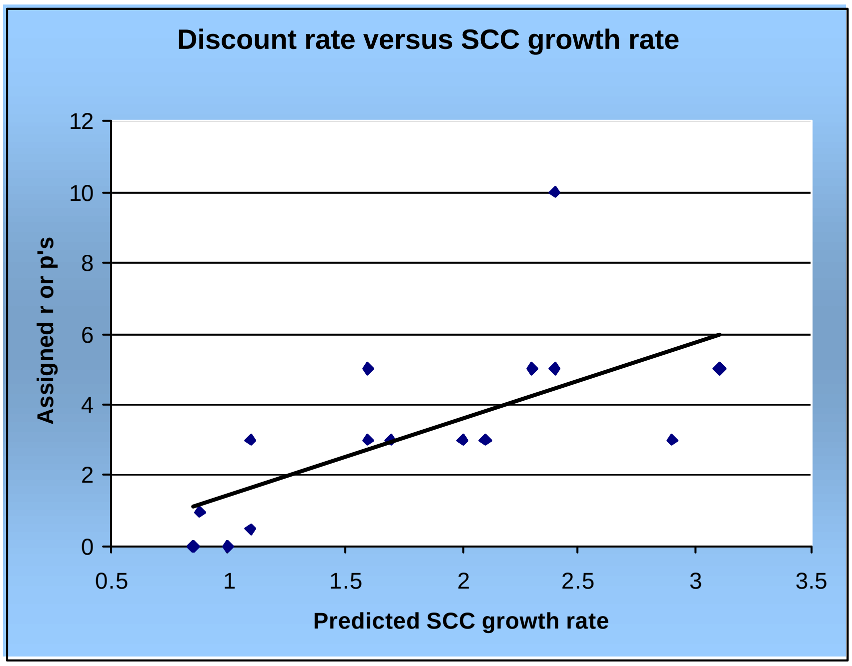


Figure 7 – Comparison between discount rates and predicted SCC growth rates listed in Table 3.

Study	r	b
Cline, 1993	0	1
	10	2.4
Eyre et al., 1999	1	0.9
	3	1.7
	5	2.4
Tol, 1999	1	0.9
	3	1.6
	5	2.3

Figure 8 – Study-specific relationships between r and b. In each of the three studies that assigned multiple r's, b increases with r (calculated from data summarized by Pearce, 2003a and Clarkson and Deyes, 2002).

In summary, net present costs from leakage estimated for a ton of injected carbon are:

$$NPC = \int_0^{\infty} L(t)D(t)SCC(t)dt$$

Substituting the following:

$$D(t) = D(t-1)(1-r(t))$$

$$SCC(t) = SCC(t-1)(1+b(t))$$

$$b(t) = \frac{w(r(t) - r(t))}{m} + \begin{cases} 0.005(1 - t/200); t < 200 \\ 0; t \geq 200 \end{cases}$$

One finds that the net present costs of leakage are:

$$NPC = \sum_0^{\infty} L(t)D(t-1)(1-r(t))SCC(t-1)[1+\frac{w(r(t)-r(t))}{m} + \begin{cases} 0.005(1-t/200); t < 200 \\ 0; t \geq 200 \end{cases}]$$

2.4 The Energy Penalty and Capture Efficiency

Until now, the analysis has assumed a perfectly efficient sequestration operation. In reality, carbon sequestration systems have significant inefficiencies. The two most important efficiency factors that need to be incorporated in an estimate of social costs are the **energy penalty** and **capture efficiency** of the system.

The physical process of capturing and disposing of CO_2 is an energy-intensive process. In order to operate a sequestration operation, power plants must burn more fuel to produce the same amount of electricity; in turn, this generates more waste CO_2 per unit of electricity. As Anderson and Newell (2003) note, "Because the capture [and disposal] process uses energy, it has a parasitic effect on electricity production (p. 7)." The energy penalty, *e*, is the fraction of production that is dedicated to powering the capture and disposal process.

Compared to a reference technology (e.g. the same facility without a storage operation), to generate the same quantity of electricity a sequestration operation must consume $1/(1-e)$ units of fuel (Ha-Duong and Keith, 2003; Keller *et al*., 2003). If 20% of a plant's electricity is dedicated to powering the capture and disposal process, then the plant must burn 25% more fuel (and hence produce 25% more carbon) per unit of electrical output.

The capture efficiency, T, is a second limitation of the sequestration operation. Modern carbon capture technologies such as monoethanolamine absorption are not 100% efficient

– generally, a significant fraction of the CO_2 escapes the capture operation. The capture efficiency, T, is the percentage of the carbon that is successfully removed.

There are at least two standard ways to address these inefficiencies; a third, which has not been used to date will be suggested here. The first approach is to ignore these variables entirely, focusing only on the disposal aspects of sequestration. This analysis examines costs per unit of *carbon sequestered*, and is therefore independent of the efficiency of the production or capture process (e.g. Caldeira *et al.*, 2001; Herzog *et al.*, 2003).

More advanced analyses incorporate the energy penalty by multiplying the cost/tC sequestered by $1/(1-e)$. This scaling factor reflects the fact that for every ton of carbon produced by the reference technology, sequestration produces $1/(1-e)$ tons to generate the same amount of electricity (e.g. Ha-Duong and Keith, 2003; Keller *et al.*, 2003; Bock *et al.*, 2002; Anderson and Newell, 2003). This approach focuses on *carbon emissions avoided* (Rubin and Rao, 2002). It ignores capture efficiency, because carbon that escapes capture is also emitted by the reference technology.

One can incorporate the energy penalty into the NPC formula by multiplying by $1/(1-e)$:

$$NPC = \frac{1}{1-e}\int_0^{\infty} L(t)D(t)SCC(t)dt$$

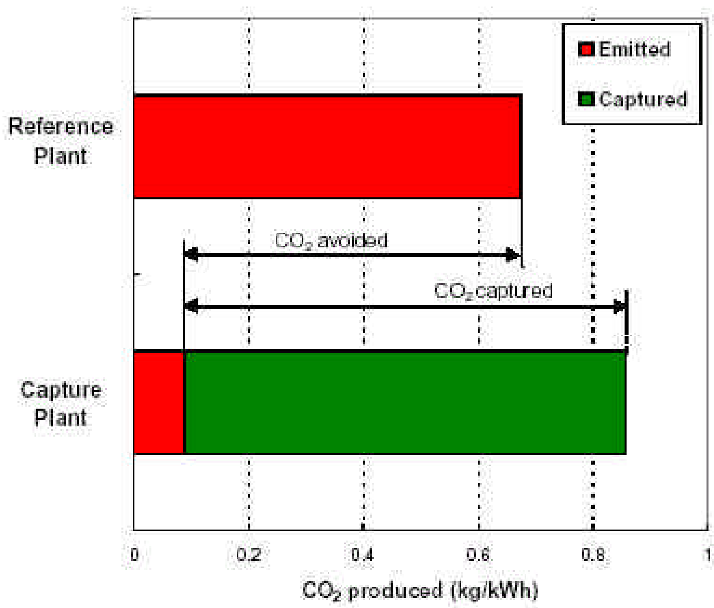


Figure 9 – Alternative methods of measuring CO_2 abatement; carbon captured versus carbon avoided (Herzog, 2001).

A third strategy—which has not been employed in models of ocean disposal to date—is to incorporate both the energy penalty and the capture efficiency directly into the cost model, thereby including the effects of both escaped carbon and sequestered carbon. This approach focuses on the *total* atmospheric costs of carbon per unit of electricity generated. There are several strong rationales for pursuing such an approach.

1) The CBA approach compares costs against benefits. With respect to electric utilities, the benefit is not *carbon storage*: the benefit is *electricity production*. As Rubin and Rao (2002) note, "Arguably, it is the cost of electricity for plants with CO2 capture that is most relevant for economic, technical and policy analyses (p. 2)." As such, social costs of sequestration technologies expressed per unit of electrical production ought to include immediate emissions.

2) The carbon-avoided method begs the question, "Avoided compared to what?"— such an approach is highly dependent upon the selection of a reference technology. Generally, the reference technology is a similar coal or gas facility (Rubin and Rao, 2002). However, the options available to generate electricity and address climate change include not only fossil fuel plants, but a range of options from energy efficiency to nuclear energy and renewables. A considerable amount of transparency is lost when comparing these technologies against sequestration on a *carbon-avoided* basis.

3) It can be misleading to exclude initial emissions of carbon. For example, Herzog *et al.* (2003) conclude that the sequestration effectiveness of mid-water disposal would be > 99.9% assuming constant carbon prices and a 3% constant discount rate. However, their stated definition of "sequestration" includes *both* capture and disposal. It is impossible to have sequestration effectiveness > 99.9% when the capture process has 10-20% inefficiencies.

The capture efficiency, T, can be incorporated by conceptualizing uncaptured CO_2 as leakage that occurs at time zero. One can multiply this initial leakage, (1 – T), by SCC_0 and the energy penalty 1/ (1 – *e*) to find the costs of the uncaptured carbon. Consistency and conservation of mass necessitate that costs of the captured carbon be multiplied by the remaining fraction, T. Thus,

$$NPC = \frac{(1-\Theta)(SCC_0)}{(1-e)} + \frac{1}{1-e}\int_0^\infty L(t)D(t)SCC(t)\,dt$$

This formulation of net present social costs yields costs relative to a unit of electricity production by the reference facility. To compare costs against non-reference technologies (e.g. mitigation), one simply has to divide the NPC by the amount of electricity produced per unit carbon by the reference technology, *K*, thereby yielding an NPC per unit electricity produced.

2.4.1 Estimating e

The energy penalty is dependent on several factors including the specific plant and removal technologies, as well as the disposal methods. Of these, the capture process and plant type are most important. A number of studies discuss likely energy penalties for future plants. For example:

· Anderson and Newell (2003) claim that in existing operations, chemical absorption imposes an energy penalty of 15-30% for natural gas plants and 30-60% for coal plants. Technology improvements can reduce penalties to 10% and 20% respectively, and to 15% for Integrated Gasification Combined Cycle (IGCC) plants.

· David (2000) summarizes the results of ten studies on energy penalties. Estimates of energy penalties in these studies range from 8.7-24% for IGCC plants, and between 9.8-16.1% for Natural Gas Combined Cycles (NGCC) facilities.

· Herzog and Golomb (In Press) suggest an average penalty of 14.6% for IGCC plants in 2000, falling to 9% in 2012. For NGCC plants, they expect average penalties to fall from 13% to 9% over the same period.

Based on the assessments above, 14% appears to be a reasonable mean estimate within a range of 8-24% depending on the operation. However, these studies only estimate energy penalties imposed by the carbon capture process. CO_2 transportation and injection, whether by pipeline or vessel, adds an additional energy penalty (Herzog, 1999). While energy requirements of disposal depend on a number of site-specific considerations such as whether the CO_2 is transported by pipe or by vessel, distance transported, and depth of injection (Summerfield *et al.*, 1993), overall, transportation is likely to be a small component of the total penalty. For example, Haugen and Eide (1996) find that a 1000 km pipeline would only impose a 3 MW penalty on a 500MW plant (0.6%). Thus, 14% appears to be a reasonable mean estimate for the energy penalty, with a broader range of 8-24% for NGCC and IGCC plants and 15-30% for traditional coal-fired plants.

2.4.2 Estimating T

There is considerably less difference in estimates of the capture efficiency. The central figure presented throughout the literature is 90% (Bock et al., 2002; David, 2000; McFarland *et al.*, 2002; Rubin and Rao, 2002), with little variation. In a recent report for the International Energy Agency (IEA), Gielen (2003) posits a direct capture efficiency of 85% in existing and likely capture technologies. He notes that "speculative" technologies employing solid oxide fuel cells might increase capture efficiencies to near 100%, but are unlikely to be available before 2030-2035.

However, apart from the carbon directly emitted during combustion, CO_2 and other greenhouse gases may be emitted indirectly during the electricity production life-cycle. The IEA (Gielen, 2003) reports that for coal-fired plants, CO_2 emitted during mining and processing represent approximately 1% of total emissions, while emissions during the transportation of the coal can range from 0-4% and the escape of methane from coalmines can add another 0-15%. Similarly, for natural gas, leakage from distribution systems can range from negligible amounts to up to 20% in the case of Russian gas in pipelines to Europe, while energy requirements of liquid natural gas production and transportation amount to almost 20% as well.

The IEA's analysis indicates that for both gas and coal, pre-combustion GHG emissions add 0-20% to baseline emissions. In support of this finding, two recent reviews have similarly concluded that the life-cycle GHG capture-efficiency of sequestration operations is approximately 85% (Davidson, 2002; Muramatsu and Iijima, 2002). If the mean figure for indirect emissions is 5%, net capture efficiency is reduced from 90% to 85%, with a broader range of 70-90%.

2.5 Chapter Summary

This chapter developed a model for estimating the NPC of atmospheric damages from sequestered CO_2. The model as currently formulated is:

$$NPC = \frac{(1-\Theta)(SCC_0)}{1-e} + \frac{\Theta}{1-e}\sum_0^\infty L(t)D(t-1)(1-r(t))SCC(t-1)(1+b(t))$$

where:

$$b(t) = 1 + \frac{w(r(t) - r(t))}{m} + \begin{cases} 0.005(1-t/200); t < 200 \\ 0; t \geq 200 \end{cases}$$

The various components of future cost—the leakage function, discount function, and social cost of carbon function—were identified, with means and distributions estimated (Table 4). The model improves upon previous work in several areas: most notably by employing declining discount rates and linking those rates with predicted growth rates in the SCC. Finally, inefficiencies in sequestration technology were explored, with the capture efficiency explicitly incorporated into a model of the economic efficiency of sequestration for the first time. The following chapter will add the previously un-assessed oceanic damages to the model. Simulations and results of the model as well as a discussion of its significance for policy will be reserved for the final section of this dissertation.

Table 4 – Estimated distributions of relevant economic variables

Variable	Central Estimate	Type of Distribution	Standard Deviation	Boundaries
L(*t*)	See Table 1	Normal	10% mean	+/- 30% mean
SCC_0	$47/tC	Lognormal	$24/tC	$12-82/tC
? G(*t*)/? *t*	(0.05%)(200-t)	Normal	20% mean	+/- 60% mean
w	0.5	Normal	0.25	0.2-1.2
µ	1.0	Normal	0.4	0.5-1.2
?(*t*)	0.25*r*(*t*)	Normal	0.125*r*	0-*r*
***r* (0-5)**	4%	Normal	2%	0-8%
***r* (6-25)**	3%	Normal	0.75%	0-6%
***r* (27-75)**	2%	Normal	0.5%	0-4%
***r* (76-300)**	1%	Normal	0.25%	0-2%
***r* (300-1000)**	0%	Normal	0%	0%
e	0.14	Lognormal	0.6	0.08-0.30
T	0.85	Normal	0.05	0.7-0.9

Chapter Three – An Empty Locker: Assessing Marine Damages from Ocean Carbon Disposal

"It is immoral to damage needlessly a remote and largely unknown assemblage of organisms—even if they are out-of-sight, out-of-mind, and apparently of little importance to the general ecological processes of the ocean—through negligent and ignorant abuse of the oceans."

-- Angel, 1982

3.1 Chapter Framework

The previous chapter on the social costs of sequestered CO_2 limited itself to examining negative effects of carbon leaked into the atmosphere. Those costs stem from carbon dioxide's role as a greenhouse gas in trapping infrared radiation. This chapter focuses on the negative effects of carbon dioxide in the marine environment, stemming instead from the acidity of CO_2 when in aqueous form. Concern about the deleterious effects of carbon disposal on the marine ecosystem has been one of, if not the single largest criticism of ocean sequestration (de Figueiredo, 2002; Fickling, 2003). However, none of the economic models addressing the social costs of sequestration include estimates of marine damages (Herzog *et al.*, 2003; Keller *et al.*, 2003; Ha-Duong and Keith, 2003). This chapter attempts to predict damages to the marine environment and integrate them into the social cost model.

The dearth of work on marine valuation, and on impacts of carbon injection in particular, makes determining marginal damages difficult. Whereas the atmospheric chapter relied on the past efforts of modelers to estimate damages from CO_2 emissions (i.e. SCC_0), the present chapter develops marginal marine damage curves from first principles. The chapter is structured in two main sections. The first section briefly examines the physical and biological impacts expected from large-scale carbon injection. The following section builds on the first by attempting to monetize likely marginal damages per ton of carbon injected into the deep-ocean.

3.2 The Effects of Ocean Carbon Disposal

Aqueous CO_2 combines with water to form carbonic acid, H_2CO_3, in equilibrium with bicarbonate, HCO_3^-, and carbonate ions, $CO_3^=$ (Figure 10). At the pH of seawater (7.7 – 8.2) the vast majority of H_2CO_3 deprotonates to form bicarbonate, while at higher pHs the bicarbonate sheds its remaining proton to become carbonate. The H^+ released in this process acidifies the water.

$$\begin{aligned} &CO_2 \text{ droplets} \longrightarrow CO_2 \text{ (aqueous)} \\ &CO_2 \text{ (aqueous)} + H_2O \longrightarrow H_2CO_3 \\ &H_2CO_3 \longleftrightarrow HCO_3^- + H^+ \\ &HCO_3^- \longleftrightarrow CO_3^= + H^+ \end{aligned}$$

Figure 10 – The carbonate system

Injecting CO_2 at 1500–2000m will create rising acidified plumes that disperse with distance. The effect on marine organisms from this carbonate acidification is not well known, and depends on numerous factors including the pH level, duration of exposure, and the specific organism and life-stage. The magnitude of pH and exposure duration depend in turn on the disposal technology and the scale of the disposal operation.

3.2.1 Disposal Options

There are three main disposal options being seriously reviewed today; stationary pipelines and tanker-carried pipelines for liquid CO_2, and dry ice tankers (Figure 1). The main thrust of Japanese research has been tanker injection at 1500–2000m, while the U.S. has focused more on stationary pipelines (Murai *et al.*, 2002). Each method creates different effects on marine life. Impacts from mid-water tanker injection will likely be limited to pelagic fauna, whereas fixed pipelines will affect both benthic and pelagic fauna. Moreover, because tankers can inject carbon over a wider area, toxicological effects will be more diluted. In contrast, dry ice tankers dropping CO_2 at depths greater than 3000m are expected to create CO_2 lakes on the ocean floor. Lake storage of CO_2 would undoubtedly exterminate most benthic fauna within the lake, while acidifying the surrounding waters to a lesser extent, depending on local currents and stability of the hydrate cap (Haugan, 1997).

The scale of disposal operation is also significant. For example, toxicological models indicate that the effects of carbon disposal from ten power plants may be one hundred times greater than effects from a single plant (Caulfield *et al.*, 1997). To date, most environmental assessments of sequestration have been theoretical reviews based on published toxicology values of the effects of 500MW coal-fired disposal (e.g. Sato, 2002; Caulfield *et al.*, 1997; Herzog *et al.*, 2000). These reviews have been complemented by a smaller number of *in situ* carbon injection experiments (Tamburri *et al.*, 2000; Aya *et al.*, 2002; Brewer *et al.*, 1999). For consistency, this chapter will concern itself with the main focus of U.S. research efforts: stationary pipeline disposal of carbon from a 500 MW coal-fired power plant at 1500–2000m.

3.2.2 Biological Effects of Disposal

At the point of injection, pH levels can fall as low as 4.0, roughly 10,000 times more acidic than natural seawater; the equivalent of lemon juice (Caulfield *et al.*, 1997). Most disposal designs entail a rising plume of aqueous CO_2 in order to dilute the acid (Figure 11). Inside a rising droplet plume, waters are still likely to reach a pH of 5.0 to 6.0 (Haugan, 1997). As the carbon-rich plume is further diluted and buffered, the pH approaches ambient conditions at a rate depending on the specific technology and surrounding currents (Herzog *et al.*, 2000). In the meantime, plumes can travel tens of kilometers away from the release site.

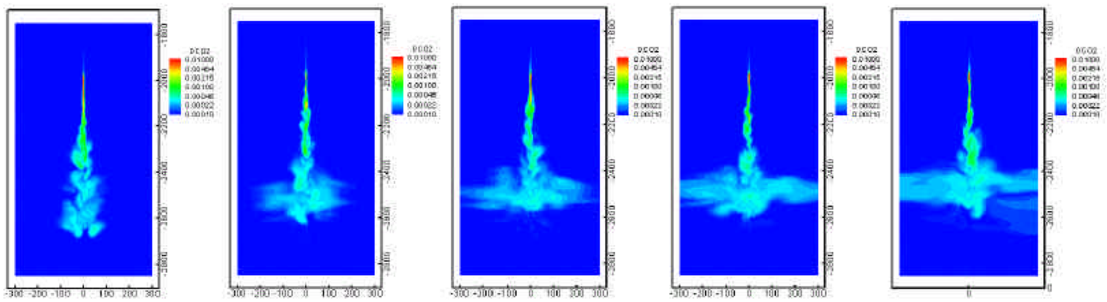


Figure 11 – Effect of rising plume dispersion design on concentration of CO_2 over a 9 hour period (Sato, 2002).

As discussed above, little information exists on the physiological effects of CO_2 on marine fauna, particularly deep-sea animals. Due to this paucity of data, the damages of acidic plumes are difficult to predict. As one pessimistic review of ocean disposal noted, "Almost nothing is known about impacts on marine organisms. To our knowledge, the few studies that directly considered the biological effects of ocean CO_2 sequestration have been toxicological models based on published values for mortality of shallow water animals when exposed to low pH solutions (Tamburri *et al.*, 2000; 95)." Most shallow water animals show increasing susceptibility to acidity with time, with smaller animals, juveniles, and larva being the most susceptible. Published data shows mortality effects which tend to become evident at a pH around 6.5 (Figure 12). While mortality is the most easily identified response, potential sub-lethal impacts include the destruction of chemosensory systems, acidosis of the blood, greater susceptibility to infections, and lower growth rates (Tamburri *et al.*, 2000).

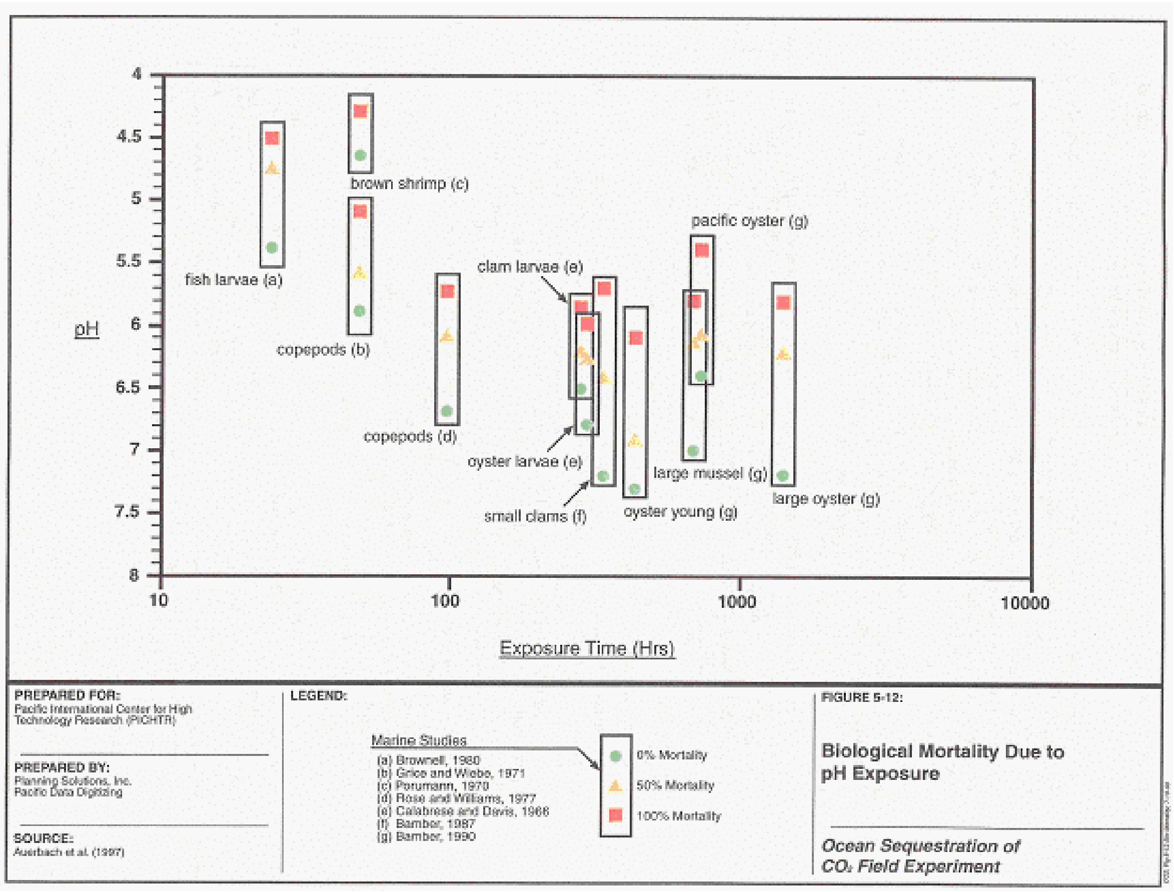


Figure 12 – Vulnerability of marine organisms to lowered pH exposure (DOE, 2001).

Deep-sea animals may be particularly vulnerable to these environmental changes for several reasons. Due to the high pressure and low abundance of food resources, deep-sea fauna are generally less active, slower growing, and longer lived than their shallow water counterparts—all traits that increase sensitivity to disturbance. Furthermore, unlike surface waters where the pH varies on a regional and seasonal scale, deep-ocean pH falls into a relatively narrow band centered around 7.8; deep-sea fauna are unlikely to have had to adapt to quick drops in pH in the past. As Haugan and Drange (1996) summarize, "The natural variability of pH at the disposal depths is smaller than in the euphotic zone, so ecosystem tolerance limits are expected to be narrower (p. 1022)."

Most studies to date predict (or assume) that the largest mortality rates will be incurred by zooplankton and benthic fauna (e.g. Herzog *et al.*, 2001). While deep-sea benthic fauna are relatively scarce compared to shallow-water fauna, they are also incredibly diverse. Around 98% of known marine species live on or in the ocean floor (Thurman and Burton, 2001). The work of Grassle and Maciolek (1992) documents levels of biodiversity (mainly polychaetes and mollusks) in muddy deep-sea sediments off of the Mid-Atlantic coast rivaling those of tropical rain forests. Deep-sea canyons, where some studies have suggested routing carbon disposal pipelines (e.g. Caulfield *et al.*, 1997; Adams *et al.*, 1995), support extremely rare populations of cold-water corals and anemones (Hecker and Blechschmidt, 1979). These benthic fauna are likely to be vulnerable to disposal operations.

Several recent environmental assessments have assumed that nekton (swimming animals) will be able to avoid the toxic plumes, thereby restricting damages to non-swimming animals (e.g. Herzog *et al*., 2001; Herzog *et al.*, 1996; Caulfield *et al.*, 1997). However, initial field experiments demonstrate that higher organisms do not necessarily avoid low-pH plumes (Tamburri *et al*., 2000). Moreover, nekton may be vulnerable at earlier life stages, or attracted to the scents of decomposing animals. Even if swimming organisms avoid the disposal site, the loss of habitat represents a harm comparable to seasonal anoxic areas found off major agricultural basins.

In addition to damages from disposed CO_2, the sequestered gases will also likely include trace gases such as SO_2 and NO_x (Herzog *et al.*, 1996). Higher acidity may also increase the production of ammonia, hydrogen sulfide, and nitrates in surrounding waters (Nelson, 2002). The effects of these chemicals, if any, have not been assessed. This analysis has only focused on acute acidification effects. Damages from chronic effects will be negligible (see Appendix 1 for calculations).

3.3 Valuation

Estimating the monetary damages from the biological impacts described above is a thorny issue. It is complicated not only by the uncertainty of the impacts themselves, but by disagreement over how to value known impacts. There is essentially no published information on the vulnerability of marine goods and services to carbon disposal. This chapter represents a first attempt at monetizing the effects of carbon disposal on the marine environment. For the purpose of organizational clarity, this discussion will compartmentalize value into two main areas, *use* and *non-use* values (Figure 13).

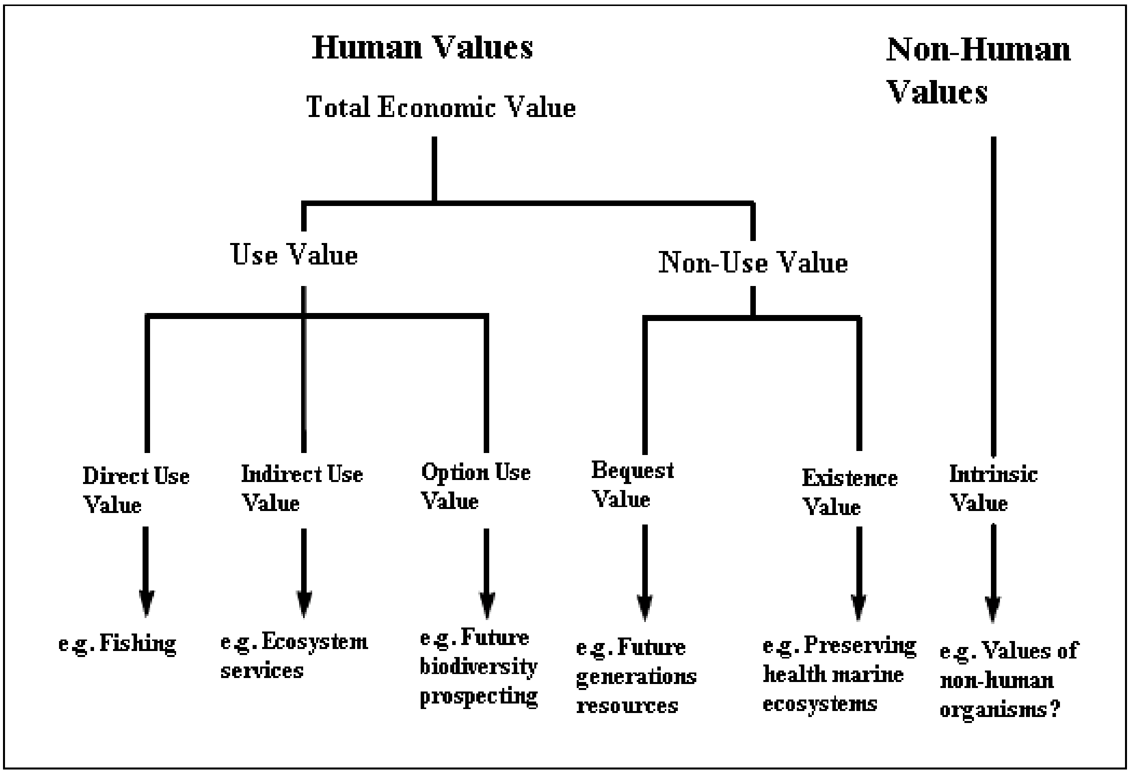


Figure 13 – Typology of total economic value for the marine environment (derived from Bateman and Langford, 1997).

Herein, use value refers to value obtained through either direct or indirect use of the marine environment such as fishing or ecosystem services. Use value also includes the *option* value, which can be conceptualized as the societal willingness-to-pay to retain the potential to use otherwise unused resources in the future; for example, the value of preserving presently unutilized genetic resources for future use. Non-use values reflect the utility obtained from the knowledge that a resource exists, independent of any use. Several models are developed herein to demonstrate that the maximum effect on use values is likely to be insignificant relative to atmospheric damages – in total, use value damages are demonstrated to be less than \$0.02/tC. In contrast, the effects of non-use value are potentially large, but also extremely difficult to estimate. Barring major work in this area, damages to non-use values ought to be treated as an independent variable with a range of illustrative values provided.

3.3.1 Use and Option Values

Before determining damages to use values it is necessary to define which uses of the ocean are vulnerable to carbon disposal. Current uses of the marine environment are manifold. They include such diverse activities as fishing, mariculture, recreation, transport, aggregates and mineral extraction, bioprospecting, and numerous ecosystem services such as waste assimilation, climate regulation, and nutrient cycling.

The majority of these uses will be unaffected by carbon disposal. Some are entirely independent of the biosphere (i.e. transportation or mineral extraction), while most of the remaining uses are limited to surface layers (e.g. recreation and mariculture) (Figure 14). Because surface waters are well-mixed and equilibrate with the atmosphere on less than a decadal time-scale, they can degas excess CO_2 over a period of a few years. In contrast, deep-ocean water (>1000m) only comes into contact with the atmosphere on a millennial time-scale (Bacastow *et al.*, 1997). As such, surface waters will presumably be less affected by direct injection (Caldeira *et al*., 2001). Those use values which appear likely to be jeopardized by mid-water ocean disposal include fisheries (particularly deep-water fisheries), genetic resources, and biologically-dependent ecosystems services.

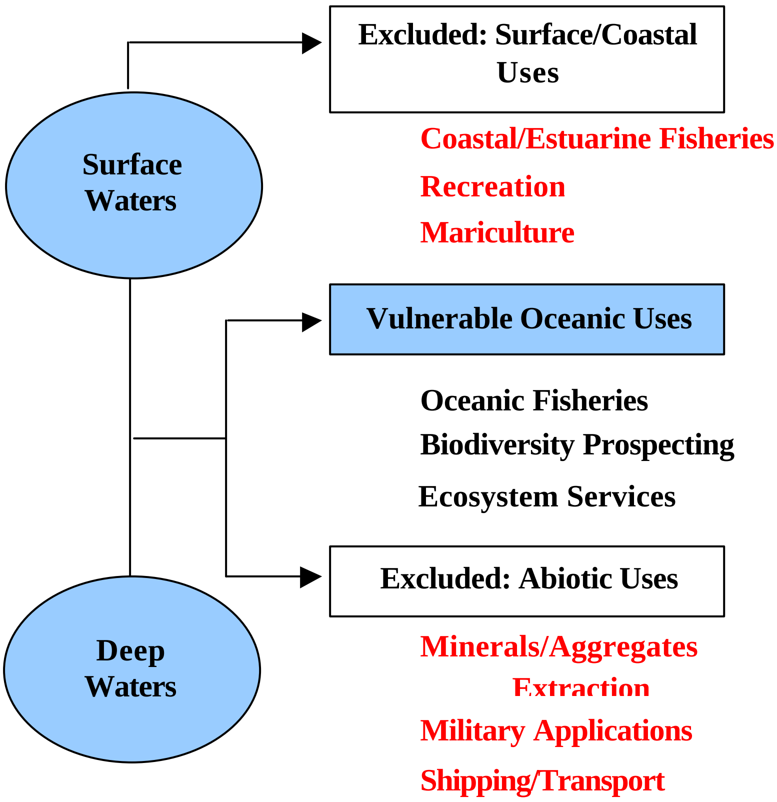


Figure 14 – Vulnerability of ocean uses to bathypelagic carbon disposal

3.3.1.1 *Estimating Damages to Oceanic Fisheries*

Estimating likely damages to a use value such as fisheries is a three-step process. One must first identify the value of the fisheries themselves, secondly the susceptibility of the fisheries to carbon disposal operations, and thirdly calculate damages on a marginal basis.

In 2000, 95 million tons (mt) of fish were harvested from the wild, for an estimated global first-sale value of $81 billion (FAO, 2002). However, $81 billion greatly exceeds the *vulnerability* of commercial fisheries to deepwater carbon disposal for several reasons. Not only does the figure include freshwater and estuarine harvests, but most commercial marine species are caught in coastal or upwelling areas, close to land (Pauly and Christensen, 1995). Given the depths of the disposal operations considered here, such shallow water species are not in danger of acidosis. Only oceanic species with a dependence on bathypelagic (1000–4000m) are likely to be vulnerable to acute acidification effects.

Of the 95 mt global catch, 8.5 mt were *oceanic* species (FAO, 2002), and most of those fish were restricted to the epipelagic (0–200m) and mesopelagic (200–1000m) zones. The FAO provides estimated harvested values for aggregated groups of species (e.g. flounders, halibuts, and soles). Summing sales of those groups containing *some* species with a likely dependency at *some* life stage on the oceanic environment (e.g. pelagic fish such as tunas and anchovies, or demersal species such as cod and flounder), a *maximum* value of $45 billion is obtained (Table 5).

Again, $45 billion is a significant overestimate. An actual value for oceanic species is likely to be closer to $7 billion (oceanic harvests multiplied by the *average* fish sale value). Even then, most commercial oceanic species are incapable of reaching the bathypelagic waters at which disposal operations are being contemplated (Table 6). While some fish live at these depths, deepwater fisheries are uncommon due to the decreasing growth rates and increasing fuel and capital costs generally associated with increasing depth. The few exceptions would include orange roughy (*Hoplostethus atlanticus*), Patagonian toothfish (*Dissostichus eleginoides*), and some lanternfish (family *Myctophidae*) which are typically located in remote areas such as the Southern Ocean.

Table 5 – Value of fisheries landings by species groups in 2000 (data - FAO, 2001).

Included Groups	2000 Landings
Flounders, halibuts, soles	– $2.3 billion
Cods, hakes, haddocks	– $6.0 billion
Misc. demersal fishes	– $4.0 billion
Salmons, trouts, smelts	– $2.0 billion
Shads	– $0.5 billion
Misc. diadromous fish	– $1.1 billion
Herrings, sardines, anchovies	– $2.6 billion
Tunas, bonitos, billfish	– $8.5 billion
Misc. pelagic fish	– $3.4 billion
Sharks, rays	– $0.8 billion
Misc. marine fish	– $1.6 billion
Misc. marine crustaceans	– $3.6 billion
Squids, cuttlefish, octopus	– $5.6 billion
Fish for reduction	– $2.7 billion
Total	**$45 billion**

Excluded Groups	2000 Landings
Carps, barbells, cyprinids	– $0.6 billion
Tilapias and cichlids	– $0.7 billion
Misc. freshwater fish	– $2.2 billion
Misc. coastal fishes	– $12 billion
Freshwater crustaceans	– $1.4 billion
Crabs	– $2.6 billion
Lobsters	– $2.0 billion
Shrimps, prawns	– $12 billion
Freshwater mollusks	– $0.4 billion
Abalones, winkles, conchs	– $0.6 billion
Oysters	– $0.2 billion
Scallops	– $0.9 billion
Clams, cockles, arkshells	– $0.8 billion
Misc. marine mollusks	– $0.6 billion
Echinoderms	– $0.3 billion
Total	**$37 billion**

Table 6 – Maximum depth of commercial oceanic species (data -Froese and Pauly, 2003).

Oceanic Species Capable of Reaching 1000 m

Species	Max. Depth
Patagonian toothfish	– 3850 m
Blue whiting	– 3000 m
Sablefish	– 2700 m
Atlantic halibut	– 2000 m
Orange roughy	– 1800 m
Antarctic toothfish	– 1600 m
Pacific halibut	– 1100 m
Monkfish	– 1000 m
Pacific whiting	– 1000 m
Grenadiers	– 1000 m

Oceanic Species Incapable of Reaching 1000 m

Species	Max. Depth
Alaska pollock	– 975 m
Swordfish	– 800 m
Atlantic cod	– 600 m
Albacore tuna	– 600 m
Wreckfish	– 600 m
Haddock	– 450 m
Blue shark	– 350 m
Round sardinella	– 300 m
Skipjack tuna	– 260 m
Bigeye tuna	– 250 m
Yellowfin tuna	– 250 m
Atlantic Bluefin tuna	– 100 m
Atlantic sailfish	– 40 m

While it is possible that disruptions in the food web could indirectly affect harvests of commercial species in surface waters—e.g. Tamburri *et al.* (2000) speculate that, "Destruction of deep-sea benthic environments by the disposal of CO_2 might lead to declines in primary production, causing an overall reduction in the biomass of higher organisms (p. 100)"—there is no evidence to suggest that such an impact will be significant. Carbon injection below the thermocline is unlikely to markedly affect surface waters, as upper ocean ecosystems are believed to have a much stronger influence on their deep-ocean counterparts than vice versa (Haugan, 1997). Moreover, commercial pelagic species are highly mobile and potentially capable of avoiding areas of adverse environmental conditions entirely. However, to demonstrate the insignificance of maximum fishery damages, these calculations will assume for argument's sake that the entire $45 billion annual harvest is vulnerable to acidification.

Given that the surface area of the Earth is 510 million km^2, 71% of which is water 3.8 km deep on average, the total volume of the ocean is calculated to be approximately 1.36 billion km^3. Hence, the maximum vulnerable production value of an *average cubic kilometer of ocean* is ($45bn/1.36bn) $33 of fish per annum.

To determine maximum fisheries damages, it is necessary to specify the minimum pH at which fisheries are likely to be affected. The U.S. Department of Energy selects a pH of 6.5 as the threshold "below which acute effects on biota *could* occur (DOE, 2001)." In a closed system, a ton of injected carbon could *at most* reduce a fifth of a cubic kilometer of water from a pH of 7.7 to a pH of 6.5 (see Appendix 2). However, the combined effects of chemical buffering and dilution reduce the magnitude of the pH drop considerably. Caulfield *et al.* (1997) determine that, in steady state, carbon from a 500MW coal-fired plant disposed in a rising droplet system would acidify 20 km^3 to pH 6.5 (Figure 15). With Caufield *et al.*'s assumed constant current of 5 cm/s, one can calculate that 1,200 km^3 are exposed to a 6.5 pH in a given year, representing a volume of ocean with an expected fisheries production value of less than $40,000/yr.

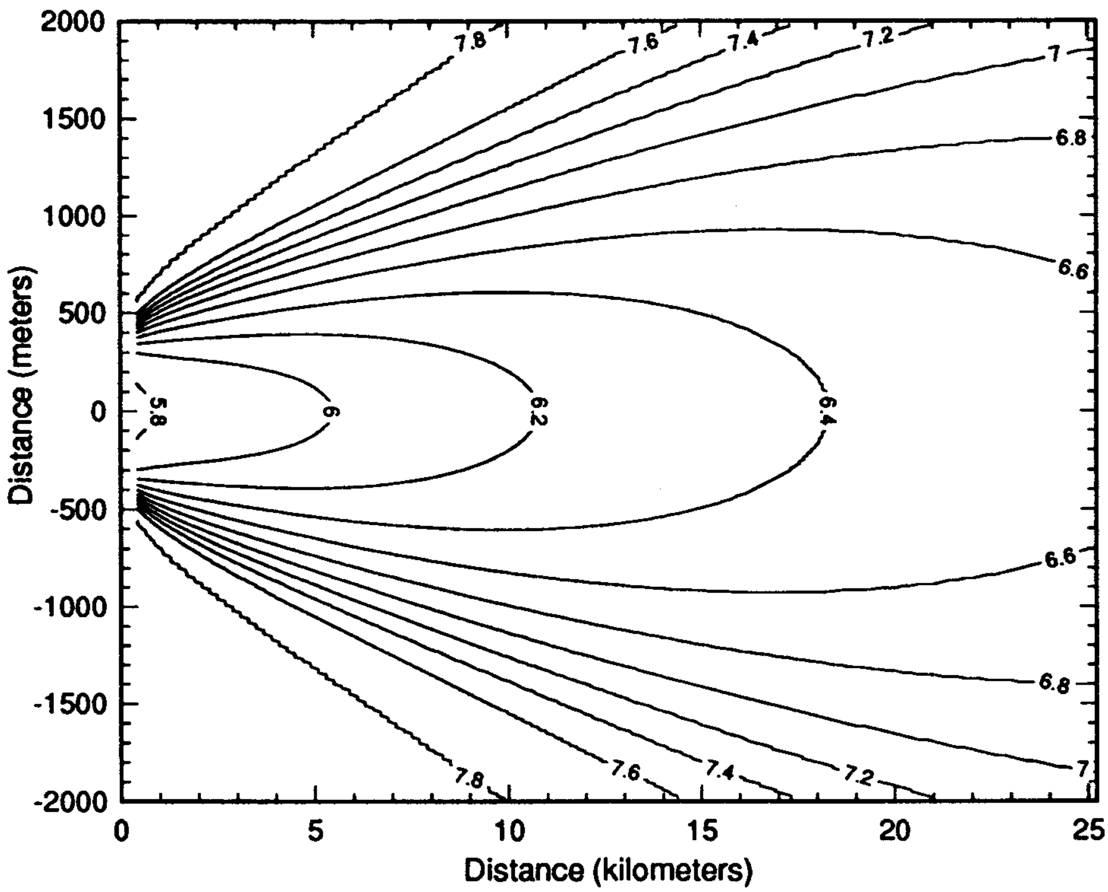


Figure 15 – pH distribution along plane of injection at 1000m depth, in steady state (Herzog *et al.*, 1996).

The duration of the acidification impact on fisheries is an important factor. If mobile fish avoid the affected area until the pH rises to above 6.5, the only damage to fisheries is the habitat loss of 20 km^3 per year, or less than $660 in production. If, in contrast, fish stocks die off in the affected waters and require an entire decade to recover, the damaged volume would be to 1,200 km^3 out of production for ten years. Assuming the value of fishery production ($33/$km^3$) rises at the discount rate, *maximum* annual damages from carbon disposal operation will range between $660 and $400,000, depending on the recovery time of the biota.

Since the sequestration operation modeled here annually disposes of 1.1 mtC (Herzog *et al.*, 1996), maximum *marginal* damages are no more than $0.0006/tC to $0.36/tC. A central marginal damage estimate (assuming a vulnerable oceanic fishery harvest of $7 billion and a one-year recovery time) is closer to $0.01/tC.

Damages to fisheries option values—or presently unutilized fisheries—are likely to be even less consequential. On a global scale, only a quarter of global fish stocks show some potential for increased utilization (FAO, 2002). Future growth in fisheries is limited by marine net primary productivity, with global landings having already plateaued over the past decade (Pauly *et al.*, 2002). More specifically, the potential for future deepwater fisheries is constrained by productivity limitations, suggesting that optimal yields from any future deepwater fisheries will be relatively modest (Pauly and Christensen, 1995). Optimists predict that harvests could be increased by 10% (FAO, 2002). Damages to fisheries option values will thus be limited to a tenth of the damages to current fisheries.

3.3.1.2 *Estimating Damages to Bioprospecting*

A less significant area of direct use is biodiversity prospecting, or bioprospecting, the field which attempts to find commercial applications to biodiversity. The genetic resources of plants and animals can have market applications in several fields, most notably pharmaceuticals and agriculture (Rausser and Small, 2000; Golin and Evenson, 2003). A noteworthy example of bioprospecting is GlaxoWellcome's $3.2 million purchase of development rights on 30,000 samples of Brazilian biota (Nunes and van den Bergh, 2001). The investigation of marine environments for biologically active agents began in the mid-1970s, as it is believed that marine organisms represent "a rich source of bioactive compounds, many from novel chemical classes compared to those found in terrestrial sources (Aalsberg, 1999; Capon, 1999)." Potential applications of these compounds include antifouling chemicals, skin care products, commercial substances such as glue and adhesives, and various agricultural products.

While bioprospecting has been heralded as a rationale and potential revenue source for terrestrial conservation, the prospects for marine bioprospecting are more limited. Research into the commercial applications of marine metabolites remains ongoing; however, few products have yet been developed. As of 1999, no biological agent of marine origin had been approved for medicinal use by the U.S. Food and Drug Administration, and only a handful of marine derivatives are seriously being investigated (Table 7).

Table 7 – Currently investigated or commercial marine bioprospecting products (adapted from Aalsberg, 1999).

Compound	**Application and Source**
Bryostatin	Anti-cancer agent derived from the Pacific Ocean bryozoan *Bugula neritina*.
Dolastatin 10	Anti-cancer agent derived from an Indian Ocean mollusk, *Dolabella auricularia*.
Ecteinascidin 743	Anti-cancer agent derived from a sea whip, *Ecteinascidia turbinate*.
Manoalide	Enzyme inhibitor isolated from the sponge *Luffrariella variabilis*.
Discodermolide	Immunosuppressant agent from the Bahamian sponge, *Discodermia dissolute*.
Didemnin B	Anti-cancer agent isolated from a Caribbean tunicate of the genus *Trididemmum*.
Pseudopterosins	Anti-inflammatory diterpene glycosides used in a skin care products to prevent skin aging; derived from the Bahamian sea whip *Pseudopterogorgia elisabethae*.
Docosahexaenoic acid (DHA)	Essential fatty acid thought useful in brain development; promoted for infant formulas/ extracted from algae *Crypthecodinium cohnii*.

With no approved pharmaceuticals and only two commercial products (Pseudopterosins and DHA), total economic benefits from marine bioprospecting to date have been small. More importantly with respect to carbon disposal, none of the species listed in Table 7 are deep-water species. As such, *current* bioprospecting efforts will be entirely unaffected by carbon injection; damages are zero.

Monetizing the damages to *future* bioprospecting is more challenging. Unlike fisheries, bioprospecting does not depend on total productivity of a species, only the existence of a large enough population to support the genetic resource. As Simpson *et al.* (1996) remark, “If all representatives of a species produce a particular compound, individuals in excess of the number needed to maintain a viable population are redundant (p. 169).” Therefore, the loss of habitat from carbon disposal does not affect bioprospecting value unless there is a concomitant loss of species.

To obtain a rough estimate for the maximum damages of carbon disposal on bioprospecting values, one can multiply the estimated average bioprospecting value of a marginal marine species by the potential to lose a viable population. Area-species curves developed in island biogeography theory offer some guidance on the amount of habitat loss necessary to trigger local extinction. Assuming a private willingness-to-pay to preserve a marginal species no greater than $10,000, a carbon disposal operation responsible for the acidification of 1,000 square kilometers of sea floor is calculated to cause less than $3,500 in lost bioprospecting option value (assumptions and calculations detailed in Appendix 3). Marginal damages to the bioprospecting option value from disposal are unlikely to be greater than $0.00002/tC. Even if one assumes the loss of 100 undiscovered species for every lost known species, damages will remain below $0.002/tC.

3.3.1.3 *Estimating Damages to Ecosystem Services*

A third component of the use value derived from the marine environment is generated by those energy flows collectively known as ecosystem services. Ecosystem services are services provided by the natural environment, including both “life-support processes” such as nutrient cycling and water-purification, as well as “life-fulfilling conditions” such as aesthetics and cultural values (Daily *et al.*, 2000). Unlike fish harvests and anti-cancer agents, these services are not saleable; as such, they are notoriously more difficult to monetize (see, e.g., Toman, 1998; Smith, 1997). To date, the most “audacious” attempt to find a global value for ecosystem services has been Costanza *et al.*'s 1997 paper in *Nature* (Heal, 1998). By aggregating estimates of the current economic value of 17 ecosystem services over 16 biomes, the authors find a mean value for global ecosystems services of $40 trillion/yr (2002 dollars).

Ecosystem valuation has been strongly criticized on several fronts including the confusion of marginal versus average values for services, and the indiscriminate scaling-up of local economic data to create global averages (Heal, 1998). However, to ignore ecosystem services entirely is to implicitly assume within the economic analysis that the services have a value of zero. Instead, rough estimates of marginal benefits from ecosystem services can serve to guide the anthropogenic use of ecosystems (Howarth and Farber, 2002). A more tenable position is to accept, as Costanza *et al.* (1998) posit in a later paper, that, "To say that we should not do valuation of ecosystems is to simply deny the reality we already do, always have and cannot avoid doing so in the future (p. 68)."

The original Costanza paper provides a useful starting point for reviewing sequestration damages to marine ecosystem services. The authors calculate a minimum value for ecosystem services from the open ocean of $308/ha yr (2002 dollars). This figure is the sum of several ecosystems services listed in Table 8.

Table 8 – Estimated value of ecosystem services in the open ocean (Costanza *et al.*, 1997).

Ecosystem Services	($/ha yr)
Nutrient cycling	143
Cultural	92
Gas regulation	46
Food production	19
Biological control	7
Genetic resources	Unknown
Climate regulation	Unknown
Waste treatment	Unknown
Habitat/refugia	Unknown
Disturbance regulation	Unknown
Recreation	Unknown
Water regulation	0
Water supply	0
Erosion control	0
Soil formation	0
Pollination	0
Raw materials	0
Total	308 + Unknowns

To determine the vulnerability of marine ecosystem services to carbon disposal, it is necessary to make two large adjustments (Figure 16). First, several of the services (food production and biological control, genetic resources, and gas regulation) are accounted for elsewhere in this review (in the fisheries, bioprospecting, and SCC sections, respectively). They are omitted herein to avoid double counting. Second, several of the marine services are unlikely to be affected by deepwater carbon disposal. These include services/characteristics fundamentally limited to surface waters, such as recreation and cultural values, as well as various terrestrial and coastal services such as flood control, waste treatment, and water regulation. The only ecosystem service classified here as vulnerable is nutrient cycling. Gas regulation is potentially vulnerable but will remain unquantified (for further discussion, see Appendix 4).

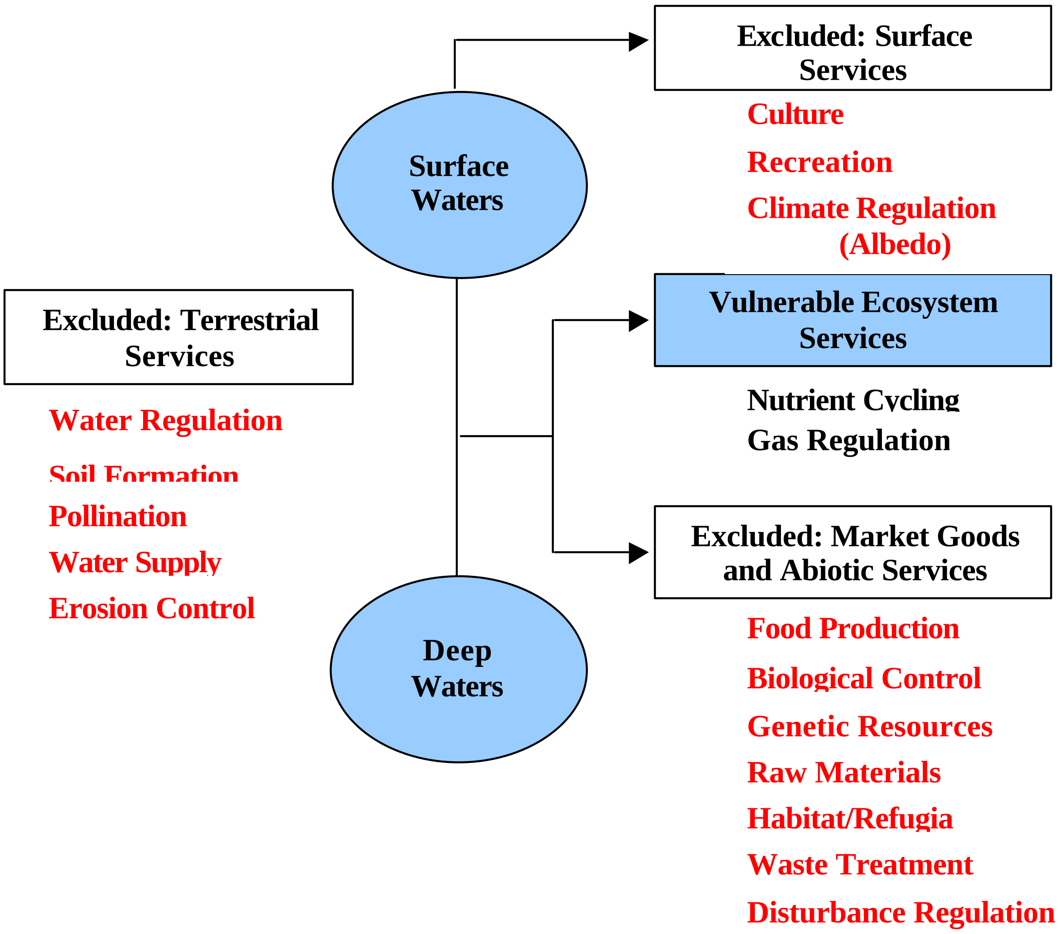


Figure 16 – Vulnerability of ecosystem services to bathypelagic carbon disposal

Nutrient cycling refers to the chemical and biological cycling of 30-40 elements within the biosphere. Nutrients are by definition critical for life-processes, and include elements such as carbon, nitrogen, and phosphorus, as well as macro-nutrients and trace elements such as potassium, zinc, and calcium. The availability of such elements partially determines the productivity and distribution of organisms, such that constant recycling is considered "essential" (de Groot *et al.*, 2002; 399). In the marine environment, the primary constituent of the nutrient cycling process is the "microbial loop," in which bacteria and viruses recycle essential nutrients, e.g. iron supplied through viral lysis (Wilhelm and Suttle, 1999). By suppressing bacterial productivity, acidification has the potential to reduce the efficacy of the loop. According to researchers at the U.S. Naval Research Laboratory, direct injection may thereby "create a substantial imbalance of the food chain and the basic elements that support the marine ecosystem (Coffin *et al.*, 2002; p. 2)."

Monetizing sequestration's effect on nutrient cycling will depend on both the marginal value of the service and the quantity lost. Costanza *et al.*'s surface area valuation of \$143/ha yr for nutrient cycling in the open ocean assumes that 1/3 of the nitrogen and phosphorus in the world's rivers is recycled offshore; the monetary value was derived from an estimated replacement value based on the costs of manually stripping nitrogen and phosphorus out of rivers.

In volume terms, \$143/ha yr averages to \$3,700/km^3 yr. However, as was the case with fisheries, surface waters appear considerably more important to nutrient cycling than deep waters. Microbial densities generally fall by a factor of five to ten over the first 500 meters of the ocean, and drop another sevenfold as one approaches abyssal depths (Fuhrman, 1999). Average microbial productivity between 1,500 and 2,000m is therefore calculated to be approximately one-third of total average productivity. This distribution reduces the nutrient cycling value to $1,200/km^3 yr at mid-water depths. Generously multiplying this number by a factor of ten in order to incorporate the cycling of other valuable elements (e.g. iron and potassium) yields a value of $12,000/km^3 yr.

The method developed to estimate the effects of lowered pH on productivity is relatively straight-forward. *In situ* observations by Takeuchi *et al.* (1997) indicate that mortality of bacteria over a four-day period occurs only when pH falls below 6.0. At any given time, a 500MW utility disposal operation acidifies less than a cubic kilometer of seawater to a pH of 6.0 (Herzog *et al.*, 1996), with an exposure duration of four days. Consequently, mortality is only predicted to be observable within the cubic kilometer nearest to the injection point. The resulting expected *maximum* damage to the nutrient cycling rounds to $0.01/tC.

In summary, though estimating a value for marine ecosystem services is not a simple task, there is a scarcity of evidence to suggest that biological activity in the deep-ocean provides humans with any significant ecosystem services. This attempt to quantify damages to known services indicates a likely maximum loss of $0.01/tC. With regard to option values, it is problematic to speculate about valuable ecosystem services not currently utilized. As such, the presumed option value is zero.

3.3.2 Non-Use Values

The second category of valuation is non-use, or "passive-use", value. Originally proposed for conservation purposes by Krutilla in 1967, these values arise independently of an individual's use of a particular resource, and are therefore often discussed as a form of altruism (Bateman and Langford, 1997). Typically, non-use values are disaggregated into existence and bequest values**,** which refer to the utility obtained from the *knowledge* that a resource is being left intact for future generations or for its own sake, respectively (Sutherland and Walsh, 1985).

Unfortunately, non-use values leave no behavioral or market trace. Seeking consensus over how to monetize them is exceedingly difficult. With respect to damages from deep-ocean carbon disposal in particular, many of the methods commonly used to estimate non-use value, such as hedonic pricing or the travel method, are simply not applicable. By default, efforts are almost entirely dependent on contingent valuation (CV) techniques, which attempt to determine value by soliciting individual willingness-to-pay (WTP) to protect a specified resource (or willingness-to-accept-compensation (WTAC) to lose the resource). The WTP approach has serious limitations in determining the passive utility derived from a resource for several reasons.

First, WTP measures are inherently limited by wealth. According to Carson *et al.* (2000), "This limitation is offensive to many who believe that government decision-making should not be based to any extent on ability to pay (p. 38)." WTP is also influenced by other factors such as culture, education, and experience. As a result, Attfield (2000) argues that "Willingness-to-pay is a poor measure of existence-value, unless existence-value is defined as willingness-to-pay; and if it is thus defined, it probably has little bearing on what ought to be taken into account in decision-making (p. 167)." Second, the preferences of future generations are not known or explicitly considered, particularly as the ability to pay changes over time (Carson *et al.* 2000). As Chapter Two illustrated WTP is expected to increase over time, making valuation of permanent damages problematic. Third, others (e.g. Rosenthal and Nelson, 1992; Weikard, 2002) have criticized the application of existence value by correctly noting that everything possesses existence value, its very ubiquity making analysis futile unless the practice is universalized.

Despite this opposition, the careful application of non-use values to economic analyses has gradually gained some acceptance. For example, in the United States, the Supreme Court ruled in *Ohio v. U.S. Department of the Interior* that non-use or existence value losses are compensable under some U.S. statutes. Similarly, the U.S. National Oceanic and Atmospheric Administration (NOAA) has allowed for non-use values as a component of compensable values (Carson *et al.*, 1994). In the UK, passive-use damages are similarly compensable if they are a component of direct use values, depending on the specific statue.

Unfortunately, *no* CV studies have been conducted specifically for deep-ocean damages, and there are few instances in which the methodology has been applied on a global scale. Perhaps the closest corollary to a carbon disposal CV study is Carson *et al.*'s (1994) exemplary valuation of the *Exxon Valdez* oil spill damages, commissioned by the State of Alaska (Duffield, 1997). The study estimated non-use damages to the marine and coastal environment between \$3.4 and \$11.3 billion (2002 dollars), based on social WTP to avoid future spills.

However, parallels with the graphic *Exxon Valdez* spill are tenuous. Americans were traumatized by the high-visibility event that involved the devastation of otherwise pristine coastal scenery and surface waters, as well as the visible suffering and death of charismatic fauna such as seabirds and marine mammals. In contrast, the invisible toxification of microbes and zooplankton two kilometers beneath the sea will presumably do less to rouse empathy. One only need glance at the current widespread degradation of marine habitat though indiscriminate fishing methods (Auster and Langton, 1999), to accept that the general public is not overly-concerned with the health of the marine ecosystem.

More generally, contingent valuation studies of habitat protection have produced a range of results. Nunes *et al.* (2000) list several contingent valuation studies for terrestrial, coastal, and wetland habitats. The mean WTPs for non-users range from $8 to over $100. Similarly, a metastudy by Loomis and White (1996) surveyed WTP studies for the protection of rare and endangered species, finding mean WTPs in U.S. households varying between $6 (striped shiner) and $95 (northern spotted owl). Again, values for deepwater oceanic habitats and fauna can only be expected to be lower, given the lack of recreational and cultural values, familiarity, information, or charismatic fauna. Similarly, the sheer distance from the deep-ocean is likely to play a role. Sutherland and Walsh (1985) document decreasing WTPs with distance for the existence value of a freshwater system in the Western United States. As deepwater marine systems are generally over a hundred kilometers from land, increasing distance should similarly reduce valuations.

Despite these arguments, there are also convincing reasons to believe that opposition to injecting carbon into the marine environment would be large. For example, benthic marine environments support high levels of biodiversity. While marine biodiversity has not been explicitly valued to date, Nunes and van den Bergh (2001) note, "…monetary value estimates seem to give unequivocal support to the belief that biodiversity has a significant, positive social value (p. 203)." Perhaps more importantly, there is empirical support for significant international opposition to marine pollution. The 1972 London Convention that prohibits the dumping of industrial waste at sea is indicative of the global commitment to protecting the marine environment. A more relevant indication is the staunch opposition encountered by an international research team's recent proposal to create a pilot carbon injection experiment to. The project was strongly eventually rejected in both Hawaii and in Norway (de Figueiredo, 2002), with opposition groups including environmentalists, fishermen, and concerned citizens galvanized over *perceived* potential damages to the marine environment. The experiment was denied despite significant international backing, scientific credibility, and a limited size and time-horizon. A full-scale disposal operation would undoubtedly raise serious concern and media attention, thereby increasing likely WTPs.

Ultimately, even if one knew regional WTPs at any given time, these do not easily translate into global WTPs, nor do they reflect marginal damages (Carson, 1998).

Estimating impacts without a serious international study to monetize damages to non-use values is purely speculative. A defensible estimate of the value might be a tenth of the *lower* boundary of the estimated *Exxon Valdez* damages, or $340 million. At $1.25 per American, this is roughly the same WTP that Loomis and White (1996) found for protecting the obscure freshwater fish, the striped shiner, and less than any other habitat protection study. Divided by an estimated 22 mt C disposed of over 20 years, the average non-use damage would be $15/tC. However, this value is pure supposition. Due to the compounding uncertainties, damage to non-use values will be treated herein as an independent variable. For illustrative purposes, the model will assume values of zero, $10, $20, $30, and $40/tC.

3.4 Chapter Summary

Chapter Three estimated damages to marine use and non-values. Damages to use and option use values such as fisheries and bioprospecting were demonstrated to be essentially negligible. In contrast, lost non-use value is potentially significant relative to atmospheric damages, but difficult to estimate due to a lack of marine CV studies. The chapter's findings are summarized in Table 9.

Table 9 – Estimated distribution of relevant marine damages

	Central Estimate	Type of Distribution	Standard Deviation	Boundaries
Use Values				
Fisheries	\$0.01 / tC	Lognormal	\$0.05 / tC	\$0 - \$0.36 / tC
Bioprospecting	\$0	None	None	\$0
Ecosystem Services	\$0.005 / tC	Normal	\$0.005 / tC	\$0 - \$0.01 / tC
Option Use Values				
Fisheries	\$0.001 / tC	Lognormal	\$0.005 / tC	\$0 - \$0.036 / tC
Bioprospecting	\$0.001 / tC	Lognormal	\$0.001 / tC	\$0-\$0.002 / tC
Non-use Value	None	Independent	None	None

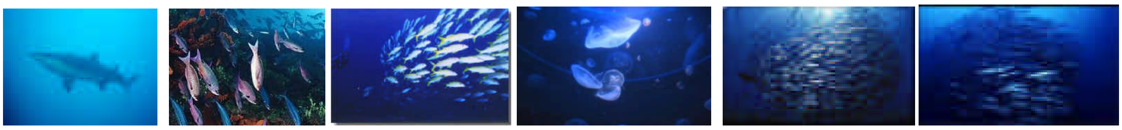
To incorporate marine damages into the atmospheric damages section, losses can be directly added into the marginal damages formula. If we define use values (U) as fishery + bioprospecting + ecosystem service damages, option values (O) as potential future fishery and bioprospecting damages, and non-use values (E) as the relevant existence value and bequest value damages, the marginal damage formula becomes:

$$NPC = \frac{(1-\Theta)(SCC_0)}{1-e} + \frac{\Theta}{1-e}(U+O+E) + \frac{\Theta}{1-e}\sum_0^\infty L(t)D_{t-1}(1-r(t))SCC_{t-1}b(t)$$

where:

$$b(t) = 1 + \frac{w(r(t)-r(t))}{m} + \begin{cases} 0.005(1-t/200); t < 200 \\ 0; t \geq 200 \end{cases}$$

Simulations and ramifications of this model are discussed in the following section.



<u>Chapter 4</u> – Results and Discussion: Davy Jones' Expensive Locker

One of the greatest pieces of economic wisdom
is to know what you do not know.
-- John Kenneth Galbraith

4.1 Chapter Framework

The model developed in the previous two chapters expresses the net present social costs of ocean carbon sequestration as:

$$NPC = \frac{(1-\Theta)(SCC_0)}{1-e} + \frac{\Theta}{1-e}(U+O+E) + \frac{\Theta}{1-e}\sum_0^{\infty} L(t) D_{t-1}(1-r(t))SCC_{t-1}b(t)$$

where:

$$b(t) = 1 + \frac{w(r(t) - r(t))}{m} + \begin{cases} 0.005(1-t/200); t < 200 \\ 0; t \geq 200 \end{cases}$$

To numerical estimate the NPC, a Monte Carlo simulation was created with Crystal Ball® decision-analysis software, with a 1,000 year time-horizon. 10,000 trials were run using the parameters specified below in Table 10 (distributions of parameters are detailed in Appendix 5).

Findings from the simulation are now presented. The results subsection details the net present costs for damages by depth, and the sensitivity of these costs to the numerous assumptions in the model. The discussion section lays out the significance of these findings in the broader context of climate policy.

Table 10 – Estimated distributions of relevant economic variables

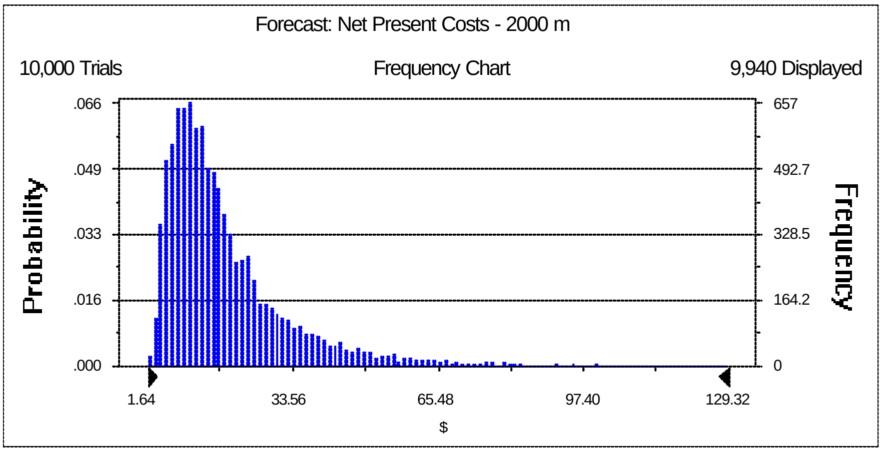
Variable	Central Estimate	Type of Distribution	Standard Deviation	Boundaries
L(*t*)	See Table 1	Normal	10% central	+/- 30% central
SCC_0	$47 / tC	Lognormal	$24 / tC	$12-82 / tC
? G(*t*)/?t	(0.05%)(200-*t*)	Normal	20% central	+/- 60% central
w	0.5	Normal	0.25	0.2-1.2
μ	1.0	Normal	0.4	0.5-1.2
?(*t*)	0.25*r*(*t*)	Normal	0.125*r*	0-*r*
r (0-5)	4%	Normal	2%	0-8%
r (6-25)	3%	Normal	0.75%	0-6%
r (27-75)	2%	Normal	0.5%	0-4%
r (76-300)	1%	Normal	0.25%	0-2%
r (300-1000)	0%	Normal	0%	0%
Energy Penalty, *e*	0.14	Lognormal	0.6	0.08-0.30
Capture Efficiency, ?	0.85	Normal	0.05	0.7-0.9
Fisheries Use Damages	$0.01 / tC	Lognormal	$0.05 / tC	$0 - $0.36 / tC
Bioprospecting Use Damages	$0 / tC	-	-	-
Ecosystem Services Damages	$0.005 / tC	Normal	$0.005 / tC	$0 - $0.01 / tC
Fisheries Option Damages	$0.001 / tC	Lognormal	$0.005 / tC	$0 - $0.036 / tC
Bioprospecting Option Damages	$0.001 / tC	Lognormal	$0.001 / tC	$0-$0.002 / tC
Non-use Values Damages	0, $10, $20, $30, $40 / tC	-	-	-

4.1 Results and Discussion

4.1.1 Results

The 2000m injection case is the main focus of this section. For a 2000m injection, mean net present costs are estimated at $21.30/tC, entailing $13.13/tC in future damages from leaked carbon, $8.16/tC from uncaptured carbon, and just $0.01/tC from damages to marine use and option values. The NPC of injection at 2000m also demonstrates a significant range (Figure 17), with a standard deviation ($40) almost twice the mean.

Figure 17 – Frequency distribution of NPC for 2000m depth, excluding non-use damages.



Moreover, the maximum NPC value in the 10,000 trials ($2,561) is more than 100 times greater than the mean, and fifty times greater than the standard deviation. This reflects significant skew in the results, indicating that the mean is affected by costly outliers, as normally associated with a gamma or lognormal distribution. As expected, a 3000m injection reduces leakage damages to $6.37/tC, while a 1000m injection increases leakage damages to \$23.57/tC; NPCs at those depths are \$14.54/tC and \$31.74/tC respectively (Table 11).

Table 11 – Components of NPC by depth, excluding non-use damages.

	Marine Damages (% NPC)	**Escaped Carbon Damages (% NPC)**	**Leakage Damages (% NPC)**	**Net Present Costs**
1000 m	\$0.01/tC (0%)	\$8.16/tC (26%)	\$23.57/tC (74%)	\$31.74/tC
2000 m	\$0.01/tC (0%)	\$8.16 (38%)	\$13.13/tC (62%)	\$21.30/tC
3000 m	\$0.01/tC (0%)	\$8.16 (56%)	\$6.37/tC (44%)	\$14.54/tC

If the NPC is divided into three components—leaked carbon, unsequestered carbon, and marine damages—the primary cause of variation in the NPC is the leaked carbon. Damages to marine use and option values are essentially constant at \$0.01/tC, with a maximum value of just \$0.22/tC, while the damage from uncaptured carbon is relatively steady at \$8.16, with a standard deviation of only \$3.90. Furthermore, both marine and uncaptured carbon damages are essentially independent of depth (at least in this analysis). In contrast, damages from leaked carbon show significant intra- and inter-depth variation. Faster leakage at 1000m increases the NPC, while the 3000m injection has lower leakage damages due to the greater influence of the discounting effect.

The analysis thus far has ignored potential damages to non-use marine values. Applying depth of injection and illustrative non-use damages as the two independent variables yields the following matrix of mean net present social costs of disposal (Table 12):

Table 12 – NPC, as determined by injection depth and marine damages.

	\$0/tC	\$10/tC	\$20/tC	\$30/tC	\$40/tC
1000m					
Leakage Damages	\$23.57	\$23.57	\$23.57	\$23.57	\$23.57
Marine Damages	\$0.01	\$10.01	\$20.01	\$30.01	\$40.01
Escaped C Damages	\$8.16	\$8.16	\$8.16	\$8.16	\$8.16
NPC/tC	\$31.74	\$41.74	\$51.74	\$61.74	\$71.74
2000m					
Leakage Damages	\$13.13	\$13.13	\$13.13	\$13.13	\$13.13
Marine Damages	\$0.01	\$10.01	\$20.01	\$30.01	\$40.01
Escaped C Damages	\$8.16	\$8.16	\$8.16	\$8.16	\$8.16
NPC/tC	\$21.30	\$31.30	\$41.30	\$51.30	\$61.30
3000m					
Leakage Damages	\$6.37	\$6.37	\$6.37	\$6.37	\$6.37
Marine Damages	\$0.01	\$10.01	\$20.01	\$30.01	\$40.01
Escaped C Damages	\$8.16	\$8.16	\$8.16	\$8.16	\$8.16
NPC	\$14.54	\$24.54	\$34.54	\$44.54	\$54.54

Clearly, non-use damages have the potential to be a significant component of total NPC, particularly at greater depths. However, it remains difficult to speculate on likely values of non-use damages barring greater work in the area.

4.1.2 Sensitivity

Rank correlation indicates that the variable with the largest influence on NPC is SCC_0 (0.57); a higher SCC_0 clearly generates a higher NPC (Figure 18). The SCC_0 is followed in sensitivity ranking by the choice of the initial discount rate, r_0 (-0.49), which reduces the NPC by discounting away future effects. The values of the two marginal elasticity terms, w and μ, also hold considerable influence over the NPC, as they largely control the evolution of the SCC. In contrast, the various marine damages (< 0.01) and the leakage variation (0.06) have little sway over results. The remaining factors—ρ/r ratio, capture efficiency, variation in damages, and energy penalty—are each significant, ranging from –0.17 to .08.

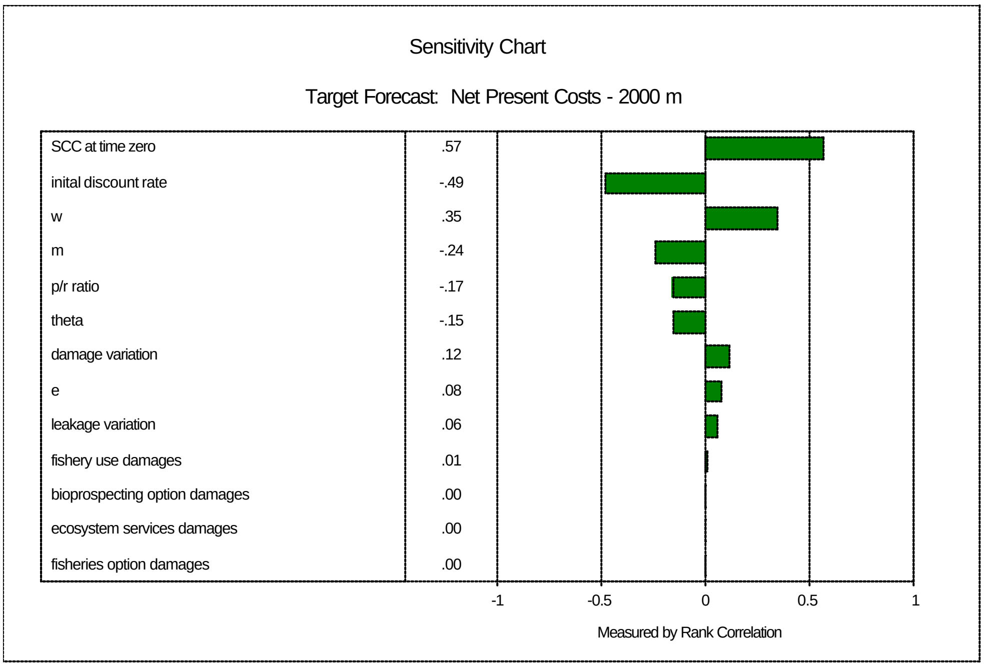


Figure 18 – Sensitivity chart of NPC at 2000m.

Though not modeled here, sensitivity to climatic parameters is an important assumption. Changing the relationship between atmospheric CO_2 concentrations growth and damage growth from linear to exponential ($\gamma = 2$) increases mean leakage costs at 2000m from \$13.13 to \$20.71/tC (+58%). If, alternatively, γ is set to 6 as Pizer (2003) suggests is possible in the case of catastrophe, leakage costs jump to \$170/tC (+1,200%). Hence, under some circumstances, delayed emissions are significantly *worse* than immediate emissions. Similarly, the debate over the pure rate of time preference, ρ, is an important one. If one sets ρ equal to zero while leaving assumptions about *r* unchanged, mean leakage damages increase 220% to \$41.99/tC.

Perhaps a more important consideration in the sensitivity analysis is the dependence of the simulation run on the IPCC S650 scenario. While the likelihood and desirability of the particular scenario may be debated, it is pertinent to recognize that large-scale carbon sequestration will entail concomitant, non-marginal effects on involuntary emissions. Significant leakage of sequestered carbon will make meeting future emissions reduction goals more difficult (and hence more costly), while at the same time reducing the capacity to meet emissions targets by diverting political attention and research funds away from non-fossil fuel alternatives.

4.2 Discussion

Three main points emerge from this analysis. They focus in turn on, 1) the lack of a substantive marine valuation, 2) the sizable atmospheric valuation, and 3) implications for future research needs.

4.2.1 Marine Damages

One of the most striking features of the results is the utter absence of use or option value that is derived from the bathypelagic marine environment. At \$0.01/tC, deepwater marine conservation can hardly be predicated on the need to protect valuable market goods. While existence and bequest values may offer some price support, there is no guarantee that individuals will be willing to pay to protect deep-ocean fauna. However, it needs to be stated that economic efficiency is not a justification for action in and of itself. To be considered justifiable, geoengineering projects need to pass *both* economic and ethical criteria. Moral concerns cannot simply be distilled into a cost-benefit analysis framework

(Howarth and Farber, 2002). Rather, philosophers such as Aldo Leopold (1949), David Ehrenfeld (1978), and more recently Warwick Fox (1995) have argued that species and ecosystems carry *intrinsic* value – something distinct from the situationally-informed existence value, and which cannot easily be monetized. Leopold's land ethic, wherein a thing is "right when it tends to preserve the integrity, stability, and beauty of the biotic community," applies equally well as an ocean ethic. It offers a code of conduct that is independent of fiscal calculus.

Similarly, the ethical issue to ocean sequestration has both sustainability and equity dimensions. A belief in the "strong sustainability" paradigm would lead one to object to ocean sequestration on the basis that irreplaceable natural capital must not be traded away for monetary assets (Neumayer, 1999): the hydrosphere ought not be sacrificed for the sake of the atmosphere, and neither can it replaced by economic growth. Others have pointed out that sequestration imposes an unfair intergenerational burden. Leakage from current injections reduces allowable levels of emissions in the future (Dooley and Wise, 2002). While humanity also benefits from the accumulated wealth, nothing guarantees that intervening generations will maintain the investments. These aspects of carbon sequestration need to be given serious attention before endorsing an ocean disposal project.

4.2.2 Atmospheric Damages

In contrast to the marine damages, the carbon leakage and uncaptured carbon damage estimates derived in this analysis are considerably *higher* than those suggested by former models (e.g. Caldeira *et al.*, 2001; Herzog *et al.*, 2003). The most obvious implication of the elevated net present social costs is the fact that it significantly impairs the economic competitiveness of ocean sequestration operations.

For a 2000m injection depth, estimated mean damages *start* at \$21/tC and rise with non-use damages. To determine *overall* costs, these social costs must then be added to private costs. Private costs for carbon disposal are significant; they are expected to increase electricity generating costs up to 50% for a gas-fired plant, and 80% for a coal utility (RCEP, 2000). According to the IPCC, a reasonable private cost estimate of CO_2 capture and disposal is roughly \$110-180/tC avoided (IPCC, 2001). More detailed analyses yield similar results. For example, Rubin and Rao (2002) find a mean private cost (\$/tC avoided) of \$165/tC for a coal-fired plant. Their 95% probability interval varies by a factor of three, from \$102 to \$271/tC. David (2000) similarly estimates private costs at \$191/tC (3.48¢/kWh) avoided for a pulverized coal plant, but notes that it is significantly less (\$99/tC avoided, or 1.72¢/kWh) for an IGCC plant. A third analysis, by Bock *et al.* (2002), employs more conservative estimates of capture costs obtained from a U.S. DOE/EPRI (2000) report on IGCC technology. The authors calculate costs for IGCC capture and 2000m disposal by pipeline or tanker at \$86 and \$141/tC avoided respectively.

A fundamental finding of this analysis is that the high private and social costs of carbon sequestration create a remarkable dilemma. To compete against fossil fuel combustion without sequestration, the SCC_0 (i.e. ideal carbon tax) needs to be sufficiently high to recoup private costs. For example, to cover private costs ($100/tC), leakage costs (an estimated mean of 0.37 SCC_0 for 2000m injection), escaped carbon costs (a mean of 0.15 SCC_0), and marine damages (say, $10/tC), the SCC_0 must be:

$$SCC_0 \geq \$100 + .37\ SCC_0 + 0.15\ SCC_0 + \$10 \geq \mathbf{\$230/tC}$$

For reference, an SCC of $230/tC is significantly outside the current range of SCC's suggested in either the Pearce (2003a) or Clarkson and Deyes (2002) metastudy. In other words, the model indicates that because of the high private costs and carbon leakage, the only scenario in which sequestration is economically viable against the reference technology is when the SCC_0 is extremely high. This alone suggests that ocean carbon sequestration is currently not economically viable.

However, as an added twist, if the SCC_0 is as high as $230/tC, the viability of marine sequestration is significantly reduced relative to *other* mitigation and adaptation options such as conservation or renewable energy. Even if the private costs associated with sequestration are subsidized, the social costs from leaked and escaped carbon can be expressed as a fraction of the SCC_0 (roughly one-half), or **$125/tC** in this instance. Given an NPC of $125/tC or greater, damages from leaked and uncaptured carbon (as well as marine damages) are likely to make sequestration an unpalatable option relative to *other* mitigation technologies.

For example, the costs of wind power (relative to pulverized coal) start at *negative* \$82/tC avoided for the best sites, and range up to \$90/tC for generation costs up to 7¢/kWh. Hydro and biofuels also offer some cost-saving opportunities relative to a reference coal-fired plant, as do building and appliance design, manufacturing, and transport (IPCC, 2001). When compared against such attractive investments, carbon sequestration is markedly less appealing. Ultimately, the economic competitiveness of marine carbon sequestration is caught in a difficult bind between the high SCC_0 required to justify the technological expenditure, and the subsequent damages from carbon stemming from leaked and uncaptured CO_2. As such, it is unlikely to be an economically viable option in the foreseeable future.

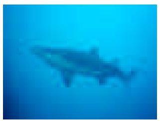
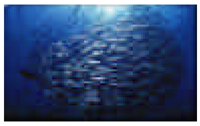
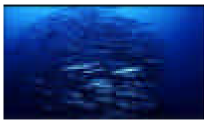
4.2.3 Future Research Needs

The third key finding of this analysis is simply the alarming paucity of data available on both environmental and economic parameters. Environmentally, the literature review has indicated that the effects of CO_2 injection on the marine environment remain poorly understood. Two key unresolved issues are the extent to which acidification will impact the microbial loop, and the duration of negative effects (depressed productivity, fish kills) on the water column. A third significant uncertainty is the extent to which sequestration will reduce the availability of carbonate ions in surface waters, thereby depressing the oceanic carbon sink and increasing CO_2 atmospheric residence time and damages.

Economically, uncertainty centers more over the evolution of the SCC(t). The growth rate of the SCC relative to the discount rate is indisputably the key factor in determining the economic efficiency of ocean carbon sequestration. However, other than the inaccurate assumptions by Herzog *et al*. (2003) and the analysis developed here, little guidance is available to modelers on the likely pathway of future carbon prices. In particular, the two marginal elasticity terms, *w* and µ, are insufficiently well known despite the fact that future willingness-to-pay to avoid climate change is instrumental in determining how damages are likely to evolve. This analysis has assumed the elasticities are constant, yet time-variable elasticities are certainly a possibility, and could have a significant effect on the NPC, particularly if *w* increases with wealth. Despite the literally *trillions* of dollars at risk from climatic damages over the coming century, there appears to be only gradual movement on estimating these crucial decision-making parameters.

Similarly, there is a patent need for additional contingent valuation studies on marine damages. The dearth of CV work for marine impacts in general is nowhere more evident than in the deep-ocean (Sumaila, 2003; personal communication). Such a monetization study should form a basic component of any future large-scale research effort into the economics of ocean disposal. Given the recent agreement between the European Union and thirteen other countries to form a “Carbon Sequestration Leadership Forum”, sufficient resources should be available to be dedicated to this important question (Doyle, 2003).

Overall, this analysis has indicated that there are serious gaps in the current understanding of long-term social costs. The substantial uncertainties involved in the economics of the sequestration process indicate that *any* assessment of net present social costs is a tenuous exercise. Given a risk-averse society and the potentially high social damages associated with carbon sequestration, the most prudent policy is undoubtedly to gather more information and focus on more certain options rather than implementing a large-scale ocean disposal strategy.



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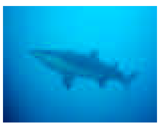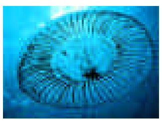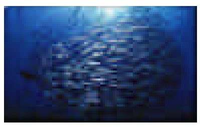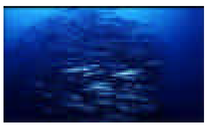

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Appendices

Appendix 1 – Chronic effects of carbon disposal

Marginal "far-field" or chronic effects stemming from a general lowering of average ocean pH are likely to be considerably smaller than acute damages on a marginal basis. While large-scale effects are highly dependent on the scale to which sequestration strategies are implemented globally, even in the limiting case of the global energy system switching to sequestration and ocean disposal, chronic effects will remain limited due to the ocean's enormous size and buffering capacity.

At an average depth of 3800m, the ocean contains over 260 times more mass than the atmosphere – society's current disposal site. As a thought exercise, one can visualize that if the atmosphere were compressed to the same density as the ocean, it would only be ten meters thick. Indeed, with an estimated 40,000 Gt C, the ocean already contains more than fifty times the carbon in the atmosphere, and four to eight times the amount believed to remain underground in fossil fuel reserves (Herzog *et al*., 2001).

In addition to its size, the deep layers of the ocean are highly unsaturated in inorganic carbon, and are geochemically buffered by carbonate sediments. It has been estimated that while pH changes of more than 0.2 units may have a detectable biological impact (Caldeira *et al*., 2001), it would require approximately 1000 Gt C (about **160 years** of current global emissions) to reduce the average pH of the ocean 0.2 units (Herzog *et al*., 2001). Hence, whereas acute effects are potentially significant, the marginal damage curve for chronic effects can be expected to be extremely flat and close to zero due to the enormous quantity of CO_2 required to reduce pH.

Appendix 2 – Maximum pH effects of injected CO_2

The amount of H^+ necessary to reduce a pH of 7.7 to 6.5 is ($10^{-6.5} - 10^{-7.7}$), or 2.97 $\times 10^{-7}$ moles/litre. At a pH of 6.5, roughly 65% of carbonic acid deprotonates to form bicarbonate and H^+, with the balance remaining as carbonic acid; essentially no carbonate forms. Hence;

$(2.97\times10^{-7}$ moles H^+/litre) / (0.65 moles H^+/mole CO_2) = 4.56 x 10^{-7} moles CO_2/litre

Converting to grams and cubic meters yields:

(4.56 x 10^{-7}moles CO_2/litre) * (44 g CO_2/mole CO_2) = 2.01 x 10^{-5} g CO_2/litre

(2.01 x 10^{-5} g CO_2/litre) * (1000 litres/m^3) = 2.01 x 10^{-2} g CO_2/m^3.

As one ton of carbon actually represents 3.67 tons of CO_2; the maximum amount of water acidified to pH 6.5 by one ton of injected C is;

(1 ton C) * (3.67 tons CO_2/ton C) * (1,000,000 g CO_2/ton CO_2) * (1 m^3 acidified to pH 6.6 / 0.0201 g CO_2) * (1 km^3 / 10^9 m^3) = 0.18 cubic kilometers acidified per ton C.

Appendix 3 – Damage estimate for marine bioprospecting option values

Damages to marine bioprospecting values /tC can be formulated as the product of the marginal bioprospecting value per marine species multiplied by the likelihood that a ton of carbon will exterminate the species.

The marginal value of marine species

Modeling the empirical costs and returns of bioprospecting in the U.S. pharmaceutical industry, Simpson *et al.* (1996) find that even under the *most optimistic* assumptions, the maximum willingness to pay to preserve a marginal species is less than $10,000, with possible minimum values as low as $0.0000005. While the $10,000 value reflects present-day WTP, if one assumes that as the gene pool is a limited natural resource and that WTP will increase at the discount rate (the Hotelling Rule), then current and future values will not diverge. Moreover, as the marine realm has fewer promising research leads than terrestrial plants, the maximum WTP per marginal marine species will be well below the $10,000 limit (Rausser and Small, 2000). This result is consistent with empirical observation: if marginal species were actually valued at several thousand dollars each, pharmaceutical companies would presumably be making greater efforts to acquire them (Simpson and Craft, 1996).

Marginal impacts to species from habitat loss

Lost bioprospecting value depends on the potential for carbon injection to render populations nonviable, i.e. extinction. Despite a surface area of 360 million km^2, the ocean contains only 235,000 known animal species and roughly another 275,000 known plant and protozoan species (Thurman and Burton, 2001; ETI, 2003); a ratio of 700 km^2 per known species. According to island biogeography theory, the number of species in a particular taxon, N, in an area of size A, is described by the formula:

$$N = j\, A^{z}$$

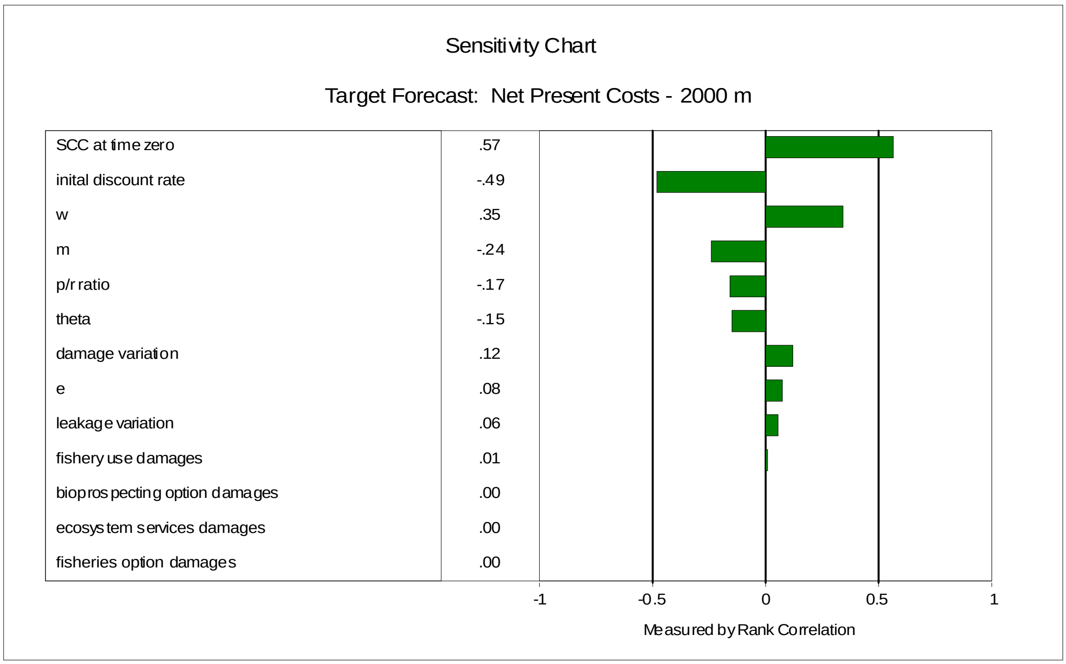
wherein *j* reflects species richness and z is an empirical constant, which for argument's sake is generously set at 0.25 (MacArthur and Wilson, 1967). With the observed area:species ratio of 700:1, the model indicates a loss of roughly *3,000* km^2 of habitat is required before the first known species is likely to be eliminated. Hence, the marginal square kilometer of ocean floor lost is only valued at $3 on average. If a 500 MW plant operates for 20 years, producing over 20 mt of waste carbon (Herzog, 1997), the marginal damages per tC are unlikely to be greater than $0.00002.

Appendix 4 – Damage estimates for marine gas regulation

Marine gas regulation is an ecosystem service that forms a major component of climate regulation. However, the impacts of sequestration on gas regulation are difficult to quantify. The fundamental effects of carbon disposal on gas regulation—i.e. adding CO_2 to the ocean, and reduced oceanic carbon uptake—are already discussed in Chapter One. However, one effect of carbon disposal on gas regulation that has not been investigated is the role that a disruption in the microbial loop could have in reducing phytoplankton productivity in surface waters. According to the U.S. Naval Research Laboratory, "Changes in the bacterial cycles can alter fundamental biogeochemical properties which must remain intact in healthy marine ecosystems (Coffin *et al.*, 2002; p. 2)." By reducing the availability of trace elements such as iron, a disruption in the microbial loop could depress productivity in surface waters, thereby affecting "the biological pump" that enhances the oceanic carbon sink. Such a disruption would increase the residence time of atmospheric carbon and further increase the rate of growth of the SCC.

However, these effects are predicated on a major disruption in microbial activity reverberating up the water column. As the nutrient cycling section indicated, microbe populations are highly robust and appear unlikely to be affected in large quantities by deepwater disposal. Moreover, the most important microbes to the biological pump are those residing in the photic zone. The connection between deepwater trophic webs and gas regulation is purely speculative, and represent an important area of future research. It is highlighted as an unknown, but not incorporated into the damages model.

Appendix 5 - NPC Model Simulation Results and Assumptions



Forecast: Net Present Costs - 2000 m

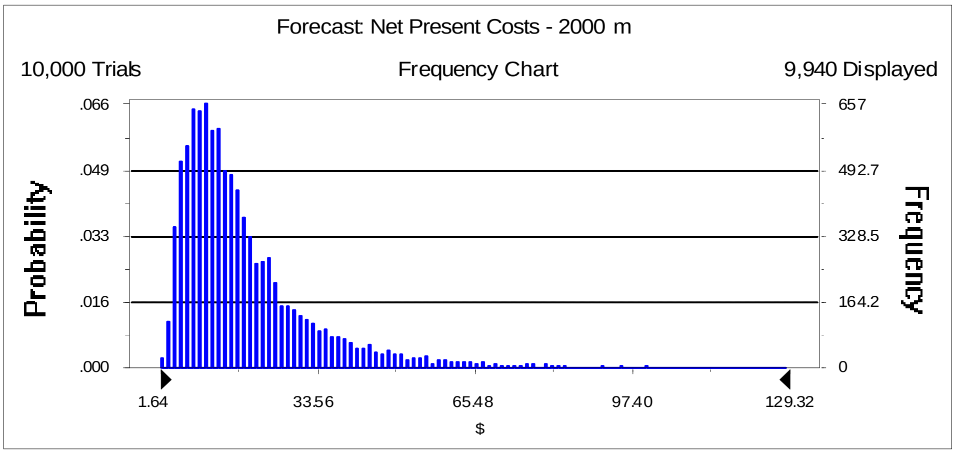
Summary:

Display Range is from 1.64 to 129.32 $

Entire Range is from 1.64 to 2,560.73 $

After 10,000 Trials, the Std. Error of the Mean is 0.40

Statistics:	Value
Trials	10000
Mean	21.30
Median	15.16
Mode	---
Standard Deviation	40.01
Variance	1,600.48
Skewness	38.27
Kurtosis	2,131.33
Coeff. of Variability	1.88
Range Minimum	1.64
Range Maximum	2,560.73
Range Width	2,559.10
Mean Std. Error	0.40



Forecast: Leakage Damages - 2000 m

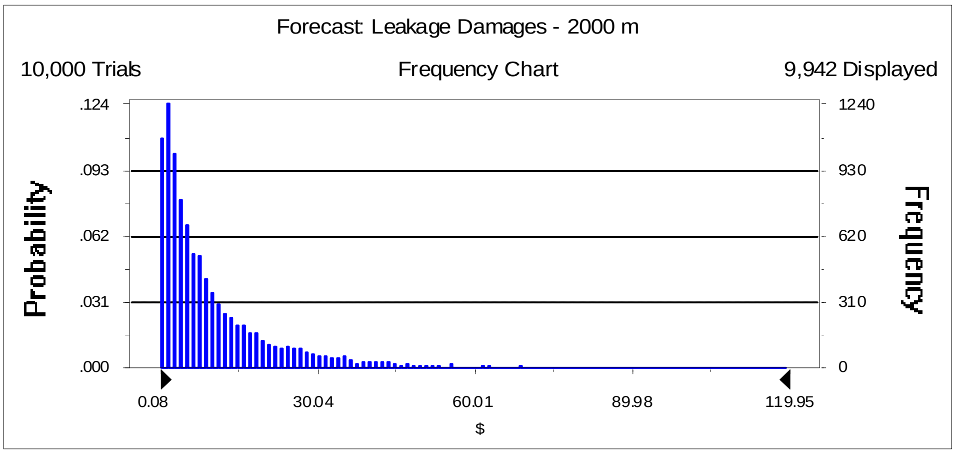
Summary:

Display Range is from 0.08 to 119.95 $

Entire Range is from 0.08 to 2,557.62 $

After 10,000 Trials, the Std. Error of the Mean is 0.40

Statistics:	Value
Trials	10000
Mean	13.13
Median	6.47
Mode	---
Standard Deviation	39.53
Variance	1,562.56
Skewness	39.73
Kurtosis	2,245.01
Coeff. of Variability	3.01
Range Minimum	0.08
Range Maximum	2,557.62
Range Width	2,557.54
Mean Std. Error	0.40



Forecast: Oceanic Damages

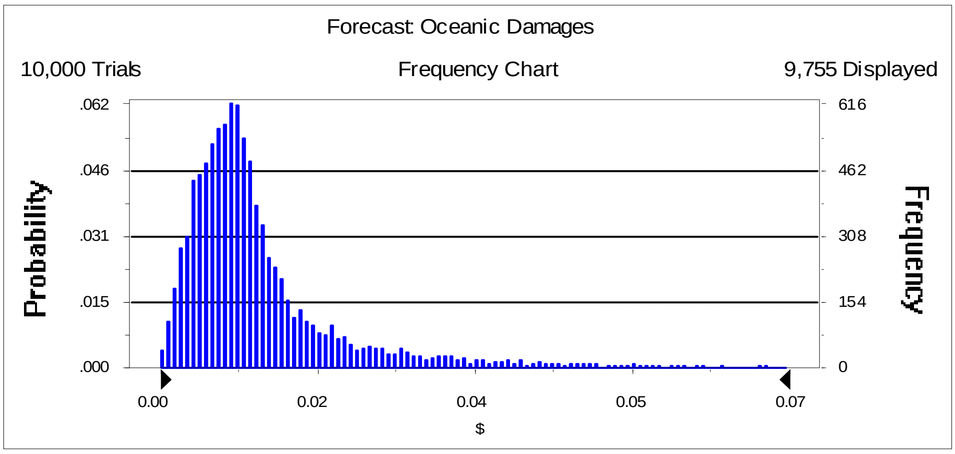
Summary:

Display Range is from 0.00 to 0.07 $

Entire Range is from 0.00 to 0.36 $

After 10,000 Trials, the Std. Error of the Mean is 0.00

Statistics:	Value
Trials	10000
Mean	0.01
Median	0.01
Mode	---
Standard Deviation	0.02
Variance	0.00
Skewness	6.62
Kurtosis	64.24
Coeff. of Variability	1.50
Range Minimum	0.00
Range Maximum	0.36
Range Width	0.36
Mean Std. Error	0.00



Forecast: Initial Damages (Escaped CO2)

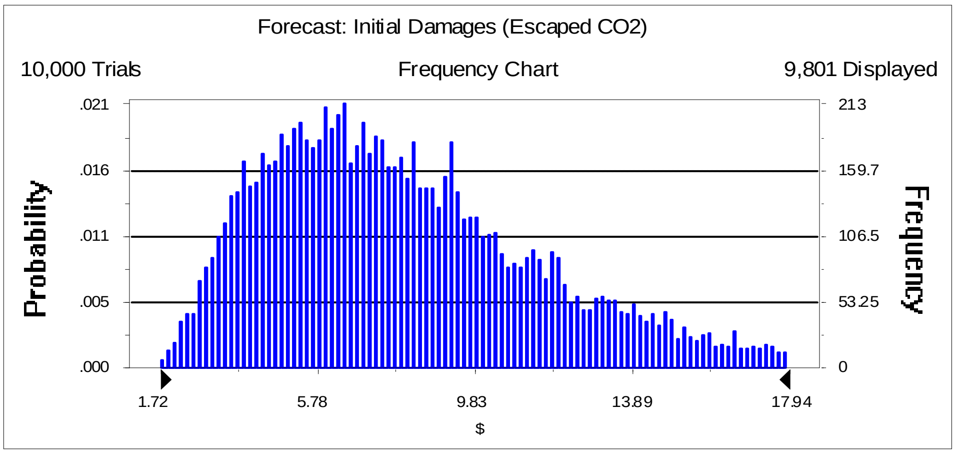
Summary:

Display Range is from 1.72 to 17.94 $

Entire Range is from 1.49 to 29.95 $

After 10,000 Trials, the Std. Error of the Mean is 0.04

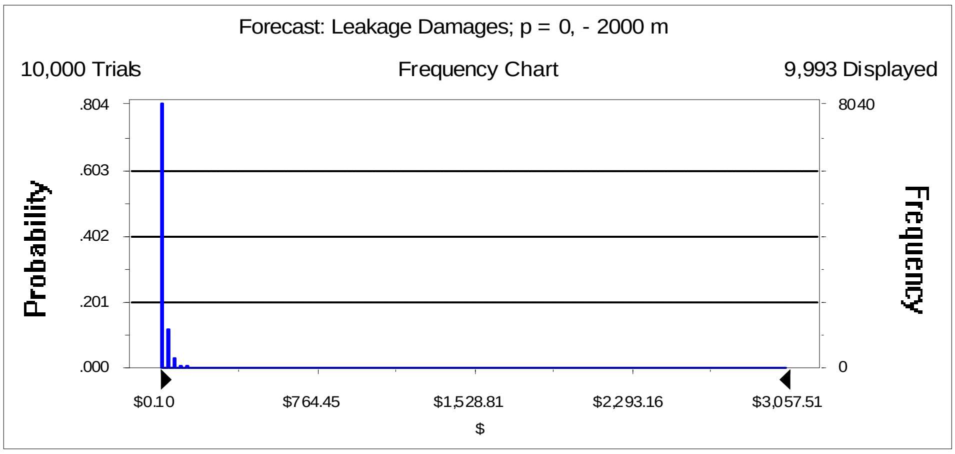
Statistics:	Value
Trials	10000
Mean	8.16
Median	7.46
Mode	---
Standard Deviation	3.79
Variance	14.33
Skewness	0.97
Kurtosis	4.02
Coeff. of Variability	0.46
Range Minimum	1.49
Range Maximum	29.95
Range Width	28.47
Mean Std. Error	0.04



Forecast: Leakage Damages; p = 0, - 2000 m

Summary:
Display Range is from $0.10 to $3,057.51 $
Entire Range is from $0.10 to $110,392.10 $
After 10,000 Trials, the Std. Error of the Mean is $11.16

Statistics:	Value
Trials	10000
Mean	$42.99
Median	$11.24
Mode	---
Standard Deviation	$1,116.19
Variance	$1,245,874.47
Skewness	96.72
Kurtosis	9,553.26
Coeff. of Variability	25.96
Range Minimum	$0.10
Range Maximum	$110,392.10
Range Width	$110,392.00
Mean Std. Error	$11.16

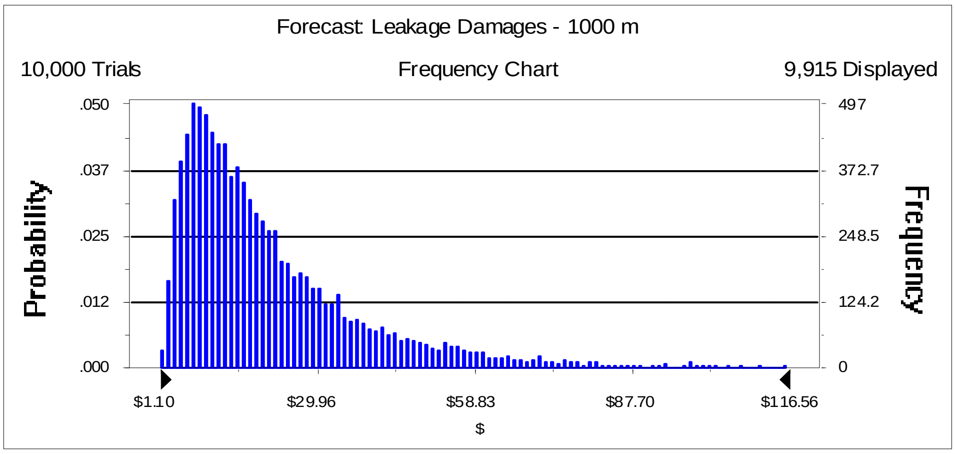


Forecast: Leakage Damages - 1000 m

Summary:

Display Range is from $1.10 to $116.56 $
Entire Range is from $1.10 to $1,909.76 $
After 10,000 Trials, the Std. Error of the Mean is $0.35

Statistics:	Value
Trials	10000
Mean	$23.57
Median	$16.61
Mode	---
Standard Deviation	$34.87
Variance	$1,215.85
Skewness	26.23
Kurtosis	1,214.35
Coeff. of Variability	1.48
Range Minimum	$1.10
Range Maximum	$1,909.76
Range Width	$1,908.67
Mean Std. Error	$0.35

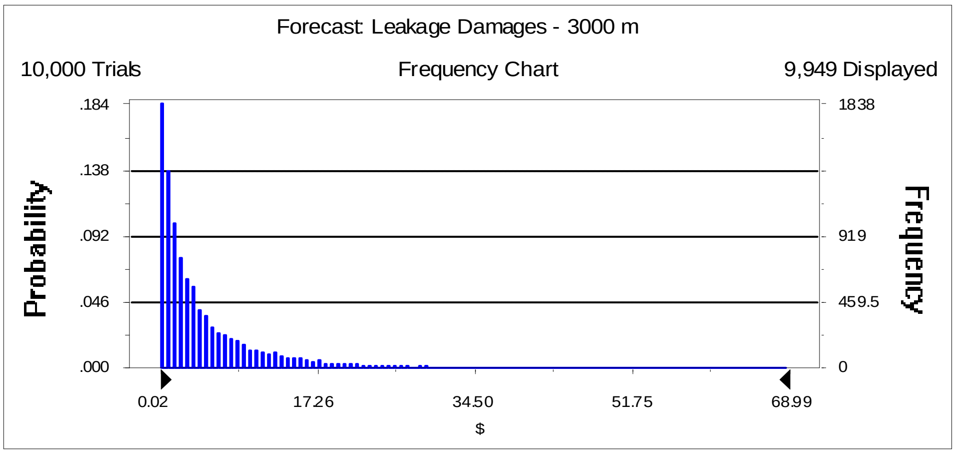


Forecast: Leakage Damages - 3000 m

Summary:

Display Range is from 0.02 to 68.99 $
Entire Range is from 0.02 to 1,555.07 $
After 10,000 Trials, the Std. Error of the Mean is 0.23

Statistics:	Value
Trials	10000
Mean	6.37
Median	2.77
Mode	---
Standard Deviation	23.09
Variance	533.38
Skewness	43.43
Kurtosis	2,572.64
Coeff. of Variability	3.63
Range Minimum	0.02
Range Maximum	1,555.07
Range Width	1,555.05
Mean Std. Error	0.23



Forecast: Leakage Damages - Gamma = 2; 2000 m

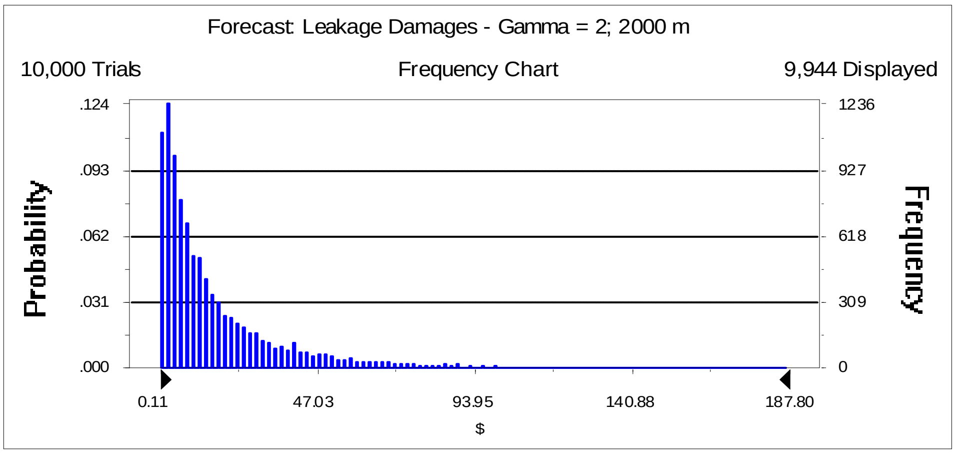
Summary:

Display Range is from 0.11 to 187.80 $

Entire Range is from 0.11 to 3,950.00 $

After 10,000 Trials, the Std. Error of the Mean is 0.62

Statistics:	Value
Trials	10000
Mean	20.71
Median	10.15
Mode	---
Standard Deviation	61.89
Variance	3,830.01
Skewness	39.02
Kurtosis	2,174.24
Coeff. of Variability	2.99
Range Minimum	0.11
Range Maximum	3,950.00
Range Width	3,949.89
Mean Std. Error	0.62



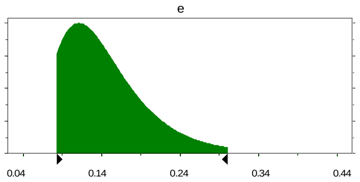
Assumptions

Assumption: e

Lognormal distribution with parameters:

Mean	0.14
Standard Dev.	0.06

Selected range is from 0.08 to 0.30

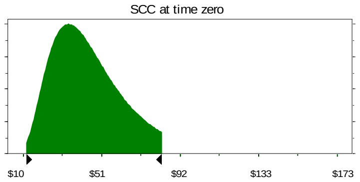


Assumption: SCC at time zero

Lognormal distribution with parameters:

Mean	$47
Standard Dev.	$24

Selected range is from $12 to $82

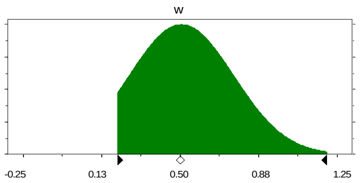


Assumption: w

Normal distribution with parameters:

Mean	0.50
Standard Dev.	0.25

Selected range is from 0.20 to 1.20

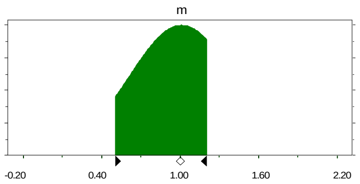


Assumption: m

Normal distribution with parameters:

Mean	1.00
Standard Dev.	0.40

Selected range is from 0.50 to 1.20

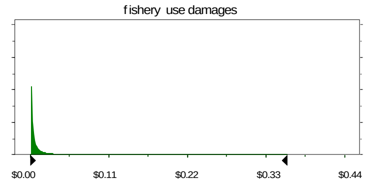


Assumption: fishery use damages

Lognormal distribution with parameters:

Mean	$0.01
Standard Dev.	$0.05

Selected range is from $0.00 to $0.36

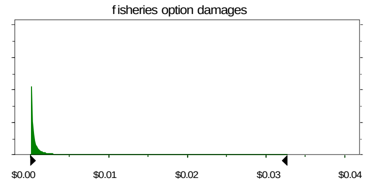


Assumption: fisheries option damages

Lognormal distribution with parameters:

Mean	$0.00
Standard Dev.	$0.01

Selected range is from $0.00 to $0.04

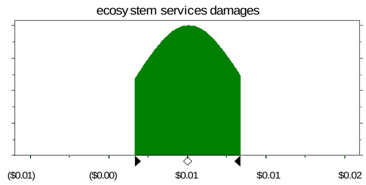


Assumption: ecosystem services damages

Normal distribution with parameters:

Mean	$0.01
Standard Dev.	$0.01

Selected range is from $0.00 to $0.01

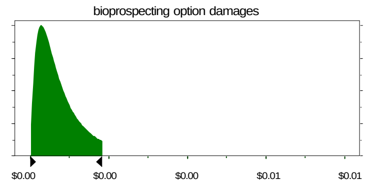


Assumption: bioprospecting option damages

Lognormal distribution with parameters:

Mean	$0.00
Standard Dev.	$0.00

Selected range is from $0.00 to $0.00

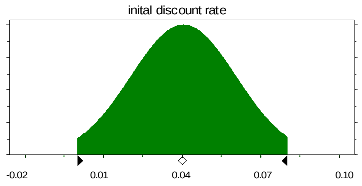


Assumption: inital discount rate

Normal distribution with parameters:

Mean	0.04
Standard Dev.	0.02

Selected range is from 0.00 to 0.08

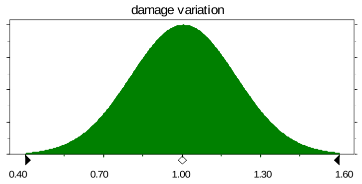


Assumption: damage variation

Normal distribution with parameters:

Mean	1.00
Standard Dev.	0.20

Selected range is from 0.40 to 1.60

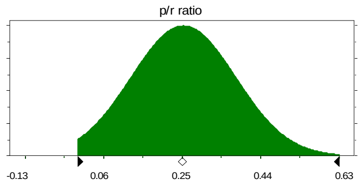


Assumption: p/r ratio

Normal distribution with parameters:

Mean	0.25
Standard Dev.	0.13

Selected range is from 0.00 to 1.00

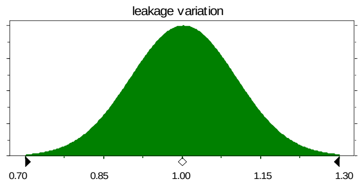


Assumption: leakage variation

Normal distribution with parameters:

Mean	1.00
Standard Dev.	0.10

Selected range is from 0.70 to 1.30



Assumption: theta

Normal distribution with parameters:

Mean	0.85
Standard Dev.	0.05

Selected range is from 0.70 to 0.90