The Costs of the Geological Disposal of Carbon Dioxide and Radioactive Waste

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Abstract Cost assessments of geological disposal of carbon dioxide and radioactive waste are presented. The scope of the cost assessments covers a range of activities from research, site identification, licensing and construction to operation, closure and post-closure monitoring of the disposal sites. The most meaningful indicator for comparison is the disposal cost per unit of electricity produced. The comparative assessment reveals important differences between the two waste products in the volume of material involved and the precautions to be taken that determine the cost per kWh indicator. The timing of investment to establish the disposal site is an important difference with significant cost implications: investments must be completed before starting CO_2 capture from fossil power plants whereas investments in radioactive waste repositories can be postponed for decades after the waste emerges from nuclear power reactors. The investment costs are significant and mid-course corrections are expensive; hence, both technologies need stable regulatory systems.

Keywords Geological disposal • Carbon dioxide • Radioactive waste • Disposal costs • Power generation costs

1 Introduction

The geological disposal of carbon dioxide (CO_2) and radioactive waste (RW) is undertaken as the final stage in the long fossil fuel electricity and nuclear power generation fuel chains, respectively. Although new clean coal technologies increasingly involve pre-combustion operations, the pathway of coal from mining to the

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power plant where it is converted into heat and/or electricity is somewhat simpler than the longer and more complex route for uranium from mines through enrichment and fuel fabrication to nuclear reactors. In the downstream phase (post-combustion of coal or burnup of nuclear fuel), the opposite is true. Depending on the carbon capture technology applied, it might take a series of chemical processes to separate and condition CO_2 into a supercritical state suitable for transport and disposal in geological formations. No such operations are needed for spent nuclear fuel (SNF). In the case of once-through fuel cycles, the most important factor between removing SNF from the power reactor and placing it in a geological disposal site is the time period required to reduce the amount of heat produced by the fuel rods before they can be packaged for disposal. If SNF is recycled, the resulting high-level RW also needs conditioning and packaging for transport and emplacement in the final repository. The costs associated with the various steps in these two fuel chains vary widely, depending on the geographical circumstances and the technologies chosen.

Quite clearly, the most meaningful approach to comparing the costs of electricity generation technologies should encompass not only the respective fuel cycles from cradle to grave but all other costs associated with building, operating and decommissioning the associated power plants. Such levelized cost estimates are regularly prepared by the OECD International Energy Agency (IEA), with the most recent update being published as IEA and NEA (2010). (This publication includes estimates of CO_2 capture costs, but not transport and disposal costs.) Complying with the mandate of this book, we limit the scope of our analysis largely to comparing the costs of the geological disposal of CO_2 and RW and to providing some insights into the related economic and financing aspects of fossil and nuclear power generation.

In all energy-economy models that include all the necessary technological and economic parameters of CO_2 capture and disposal (CCD) options in their technology dataset, the costs of reducing greenhouse gas emissions from fossil fuel electricity by capturing CO_2 and disposing of it in geological formations are implicitly compared with all other energy supply technologies. The resulting energy mix from these models represents the least-cost supply portfolio in which fossil energy sources, with and without CCD, and all other energy sources and technologies are utilized under a given set of external assumptions (about resource prices and availability, technological performance and costs, etc.) and constraints (e.g. CO_2 emission ceiling).

Some studies explicitly compare the costs of selected technologies to reduce CO_2 emissions from the energy system. Rogner et al. (2008) analyse the cost-effectiveness of two main baseload power technologies: fossil-based electricity with and without CCD and nuclear power in nine countries. They calculate life cycle electricity costs in present value terms from national energy studies comprising country-specific technological and economic data. The authors find that adding CCD results in a considerable increase in the cost of fossil fuel electricity (mainly coal). Relative to the reference cases, this would completely eliminate the cost advantage of fossil-based electricity without CCD over nuclear power in some of the countries analysed (Argentina, Bulgaria, China, India) and significantly increase the cost advantage of nuclear power in other countries included in the study (Korea, Pakistan, Poland, Romania, Russia).

Tavoni and van der Zwaan (2009) use the global integrated assessment model WITCH (Bosetti et al. 2006) to compare the competitiveness of coal-based electricity with CCD and nuclear power. They find that, using the standard formulation and assumptions of the WITCH model, but removing the expansion constraints usually adopted for nuclear power in most energy-economy models (to reflect perceived limits due to public acceptance, political and other non-economic reasons), global nuclear power capacity is projected to expand by 15–20 GW/year under a target of stabilizing global greenhouse gas emissions at 550 ppm CO_2-equivalent, while the expansion rate is projected to increase from about 20 GW/year in 2020 to almost 35 GW/year in 2050 under a 450 ppm stabilization scenario. The authors suggest that, provided public acceptance and politics do not prevent the expansion of nuclear capacities to their full economic potential, considerable improvements will be needed in CO_2 capture costs and technological efficiencies if CCD is to compete with nuclear power and provide a large share of CO_2 mitigation.

Framing a meaningful comparison of the disposal costs of CO_2 and RW is difficult for various reasons elaborated in preceding chapters: the volumes and physical and chemical properties (Bachu and McEwen 2011), the human health and environmental hazards posed (West et al. 2011), the transport (Gómez and Tyacke 2011) and site engineering (Tshibangu and Descamps 2011) technologies of CO_2 and RW heavily influence the various cost components and thus the total disposal costs. Relating these disposal costs to the amount of electricity generated, to the total fuel cycle costs and to the levelized electricity costs could provide information about the similarities and differences between the waste disposal cost profiles of the two technologies; however, the unavailability of consistent and comparable data makes the comparison largely unreliable.

An additional problem with comparing costs in both technologies over time is the general escalation of all technology costs (all constructions and equipment for CO_2 and RW disposal) over the past 8 years. The reviews presented in this chapter provide the original values from the cited studies but for the summary and comparison tables a common metric of 2010 US$ is adopted by applying the appropriate GDP deflators and converting other currencies at average 2010 exchange rates.

Section 2 of this chapter presents recent estimates of the costs of the geological disposal of CO_2, followed by a similar overview of RW disposal cost estimates in Sect. 3. Based on the issues raised in these assessments and on the resulting cost estimates, Sect. 4 provides a comparative assessment, followed by a concluding summary in Sect. 5.

2 Costs of the Geological Disposal of CO_2

This section provides a short overview of recent studies on the cost of CCD. First, the main cost elements are itemized. This is followed by cost estimates published over the past few years. The sample of studies involves rather different approaches ranging from specific disposal case studies (project costs) to aggregate supply curves of disposal potential. Care is needed when drawing general conclusions from such a diverse sample. Finally, the relative importance of the disposal costs in the fuel cycle and levelized electricity costs are analysed.

2.1 Cost Elements

The three main phases in the CCD chain include capture, transport and disposal. The cost estimates for all three steps are presented in the next section, but here the focus is on the disposal phase. This phase starts at the point of CO_2 delivery to the disposal site which, in most cases, is the end of a pipeline.

The first cost items include the preparatory steps in the site establishment process that starts with geological screening of potential sites, site exploration and characterization (geological, geophysical and engineering feasibility analyses) that lead to site selection. This is followed by reservoir characterization and evaluation. Licensing and stakeholder engagement also involve costs. The bulk of the disposal costs arise from the site construction: design and drilling the injections well, mounting in-field pipelines if needed, installation of the surface facilities: scrubber (to remove residual liquids), compressor and other equipment. Infrastructure costs may arise from road construction and energy supply connections. Operating costs cover labour, maintenance and energy.

Most cost estimates published to date include the above site characterization, capital and operating costs, but ignore other, potentially important cost items. Monitoring costs accrue throughout the whole process from establishing the baseline monitoring system through the post-closure monitoring of the disposal site, possibly over decades, but they are usually not included in the cost estimates. It is worth noting that monitoring costs are also interlinked with other cost items. A system consisting of many small reservoirs that are close to sources of CO_2 will have higher monitoring costs than a system with a few large-scale reservoirs that are connected by a well developed pipeline system. Longer pipelines add to the cost of the system, but the cost can be offset by lower disposal costs in large saline aquifers. The difference in monitoring costs (depending on how far out in time those costs are counted and the discount rate used) can tip the scale in favour of a pipeline network with a few major disposal sites.

Additional cost items might arise from remediation and liability. Remediation measures might be required when leakage occurs. Additional expenditures may occur to address long-term liability issues. These items are difficult to estimate and are excluded from most cost estimates published to date.

In the absence of industry-scale CCD experience, cost estimates draw on industrial analogues related to the relevant technological components. The main source is the oil and gas industry with its well drilling and injection technologies and associated knowledge base on monitoring and modelling of reservoirs. Enhanced oil recovery (EOR), acid gas and other liquid waste injection projects as well as experience with underground natural gas storage sites provide a good basis for cost assessments.

A comprehensive technology and cost analysis for geological CO_2 disposal is presented by the US Environmental Protection Agency (US EPA 2008). The report presents a general costing methodology together with the costs of specific technologies and operating practices that might be adopted for CO_2 disposal costing studies. Unit costs are estimated in terms of cost per site, per well, per unit of area or other relevant characteristic, depending on the nature of the cost item. The EPA report includes the following cost categories:

- Geological site characterization;
- Monitoring;
- Injection well construction;
- Area of review and corrective action;
- Well operation;
- Mechanical integrity tests;
- Post injection well plugging and site care;
- Financial responsibility;
- General and administrative.

The principal difficulty in all estimates is that disposal costs vary widely depending on the geological characteristics of the site. The depth, thickness and permeability of the host geological formation determine both the construction (number of wells, in-field pipelines) and the operation (injection) costs. Location is another key cost driver: offshore geological disposal is significantly more expensive than onshore disposal. (Offshore deep ocean disposal is not considered in this chapter.) Hence cost estimates need to be handled with care: extrapolation from one site to another could be rather imprecise or outright wrong.

2.2 Cost Estimates

The most expensive item in the CO_2 capture–disposal chain is capture. The cost of capture is calculated and presented in the literature in two ways: one is based on incremental cost of the capture equipment and its operation (a simple engineering cost) while the other takes net cost and emissions of a plant with capture equipment and compares this to a reference plant without capture (an economic cost). Accordingly, the costs quoted below include both types: incremental engineering as well as net avoided costs. Comparing these costs to each other is difficult because there is not sufficient information available to convert these estimates to a common metric, since the ratio of capture and avoided costs varies by 10–40% between different projects. Yet even with this caveat, ballpark comparisons are still meaningful.

The starting point for our cost review is the report by the Intergovernmental Panel on Climate Change (IPCC 2005) which summarized the then current state of knowledge about CCD. Chapters 3–7 of the IPCC report provide detailed assessments of the technologies involved, environmental, monitoring, risk and legal aspects, as well as the cost estimates for CCD phases, from capture and transport to disposal in underground geological formations, the ocean at great depth and via mineral carbonation. Chapter 8 on costs and economic potential (Herzog et al. 2005) draws on these chapters and summarizes a large number of the studies that were then available.

Not surprisingly, most of the attention of the studies assessed by Herzog et al. (2005) focuses on CO_2 capture costs, which are by far the most expensive component of the capture, transport and disposal chain. The authors find the following cost ranges for CO_2 capture expressed in US\$/t CO_2 captured (that is simple engineering costs) for newly built plants (all in 2002 US\$): 33–57 for new natural gas combined cycle (NGCC) plants, 23–35 for pulverized coal (PC) plants and 11–32 for integrated gasification combined cycle (IGCC) plants. The main factor determining the cost of CO_2 transport via pipelines is the volume transported (see also Gómez and Tyacke 2011). For a mass flow rate of more than 15 million tonnes (Mt) CO_2/year onshore on 'normal' terrain (low population density and no high mountains), the cost range is between US\$1 and US\$2/t CO_2 for a distance of 250 km.

In addition to site-specific features, the crucial factor determining the disposal costs is whether the CO_2 can serve a revenue generating activity through EOR, enhanced gas recovery (EGR) or enhanced coalbed methane (ECBM) recovery, or not. Onshore disposal with EOR can generate a revenue of US\$10–16/t CO_2, depending on the prevailing oil prices. The cost range for disposal in saline formations and depleted oilfields is wide: US\$0.5–8.0/t CO_2. Cost estimates for monitoring are in the range of US\$0.1–0.3/t CO_2 (Herzog et al. 2005). The IPCC report (IPCC 2005) also estimates the costs of disposal in deep oceans, but this option remains controversial because of environmental concerns, and it is more expensive than the geological alternatives. At the high end of the spectrum, mineral carbonation cost estimates are in the range of US\$50–100/t CO_2.

The IPCC report (IPCC 2005) identifies many factors that make the comparison of CCD cost estimates difficult. They include technology- and location-specific factors, the differing boundaries of the capture and disposal system and the different metrics adopted: the investment costs for the capture system, the incremental product costs (e.g. cost of electricity), the cost of CO_2 avoided and the cost of CO_2 captured and removed. Some studies also add a temporal dimension by assuming different learning rates resulting in declining unit costs over time. These difficulties are also involved in comparing the estimated or projected costs of the whole or of various components of the CCD system reviewed in this section.

The focus of the CCD studies published since the IPCC report remains on the capture phase. The studies cover a wide range of issues such as: retrofitting options for existing coal-fired power plants (e.g. Korkmaz et al. 2009), their operability with CO_2 capture equipment installed that is significantly different from their normal design conditions (see Alie et al. 2009), and the environmental impacts of the processes and materials involved, like the amines used in post-combustion capture of CO_2 (Eide-Haugmo et al. 2009). Large utilities report on their CCD pilot projects (Renzenbrink et al. 2009; Strömberg et al. 2009). As capture costs represent the largest share in the overall CCD costs, many studies are devoted to them (e.g. Ho et al. 2009). In addition, assessments of CO_2 disposal capacities in many countries and world regions attempt to provide more precise estimates (see the report by Vangkilde-Pedersen et al. (2009) about the European Union (EU) GeoCapacity project).

The number of studies reporting CCD cost estimates has been increasing since the publication of the 2005 IPCC report. Most of them focus on the capture component and omit transport and disposal costs altogether or add these items as lump sum figures from other sources. A selected set of cost studies are summarized in the paragraphs that follow. Table 1 summarizes the results of the cost estimates.

McCoy and Rubin (2005) review the CCD literature and find 'frequent inconsistencies and lack of clarity in defining the scope of the CO_2 capture, transport and [disposal] components'. They present engineering and economic models of CO_2 transport via pipelines and geological disposal of CO_2 in deep saline aquifers. The aim of their models is to provide first-order cost estimates that are sensitive to site-specific or project-specific technical and financial parameters. The authors analyse a case study in the Wabamun Lake area of Alberta, Canada. Their results indicate significant uncertainties in the cost of CO_2 transport and disposal, primarily due to the variability of the geological parameters of the reservoir, as well as to other factors such as transport distance and power plant capacity factor. McCoy and Rubin (2005) find that the combined cost of transport and disposal (on a cost per tonne CO_2 basis) could represent more than 32% of the total CCD cost, as opposed to other estimates of less than 15%. The case they analyse involves CO_2 disposal from a 500 MW coal-fired power plant. The median cost for CO_2 transport is US$0.44/t, US$1.44/t for disposal, and US$1.94/t CO_2 for the combined median cost of transport and disposal. The authors find that disposal costs are more variable than transport costs, and that the total cost varies between a fifth percentile of US$0.78/t and a 95th percentile of US$14.59/t. Based on these results, the median cost of transport and disposal seems to be a small part of the total cost of CO_2 disposal, but according to their study there will be cases in which the cost of transport and disposal are large.

Vosbeek and Warmenhoven (2007) provide a comprehensive assessment of the opportunities and prospects for CCD in the Netherlands. They conclude that retrofitting existing power plants with CO_2 capture equipment would reduce their efficiency to an unacceptably low level. They analyse three business concepts: stand-alone (one CO_2 source with a short pipeline to a dedicated disposal site), network (several sources with a pipeline network to several disposal sites) and a network with CO_2 utilization (for EOR or other industrial purposes). The report calculates the integral costs of capture and disposal from the costs per tonne of CO_2 emissions avoided in 2006 €. Based on an Ecofys report (Hendriks et al. 2007), the authors derive total CCD costs of €29 and €39/t CO_2 for the stand-alone and network cases, respectively. The transport costs are estimated at €1/t CO_2 for the stand-alone case (one plant located 40 km from a good disposal site) and €4/t CO_2 for network projects, while the disposal cost in both cases is taken to be €2/t CO_2. When CO_2 is used for EOR in the North Sea, transport costs are assumed at €5/t CO_2 and disposal costs range from −€3 to −€22/t CO_2, depending on the amount of oil yield and assuming an oil price of €20/barrel. Accordingly, the total CCD costs vary between €16 and €35/t CO_2.

Table 1 Overview of CCD costs estimated by selected studies

Source	Capture	Transport	Disposal	Total	Notes
IPCC (2005)	NGCC: 33–57 PC: 23–35 IGCC: 11–32	1–2 (15 Mt CO_2/year, 250 km)	−10 to −16 (EOR) 0.5–8 (saline formations) 50–100 (mineral carbonation)		All in 2002 US$/t CO_2 captured
McCoy and Rubin (2005)		0.44	1.44	1.94	US$/t CO_2; median transport and storage costs, total excludes capture costs
Vosbeek and Warmenhoven (2007)	26 for stand-alone 33 for network project	1 for stand-alone 4 for network project	2 −3 to −22 with EOR	29 (stand-alone) to 39 (network) 16–35 with EOR	€/t CO_2 captured
Ploumen et al. (2007)	IGCC: 25–30 PC: 32–42 PC retrofit: 40–52				€/t CO_2
Hendriks et al. (2007)		1.0–1.6			20–10 Mt CO_2/year
Pöyry Energy Consulting (2007)	21–27 for coal retrofit 31–35 for gas retrofit	1–9	Up to −16 with EOR, 1 in aquifers, 1–20 in oil- and gasfields	21–44 (for a range of 10–180 Mt CO_2/year)	£/t CO_2 abated
IEA (2008)		2–6 for 2 Mt/year to 100 km	10–25		US$/t CO_2 captured
Captured:					
NGCC	50–90	1–3 for10 Mt/year to 100 km			
PC and IGCC	40–55				

Avoided:						US$/t CO_2 avoided
NGCC	60–110 in 2010					
	55–90 in 2030					
PC and IGCC	60–75 in 2010					
	50–65 in 2030					
BCG (2008)	25	2–3		4–5	31–33	€/t CO_2
Hamilton et al. (2008)	52		10		62	US$/t CO_2 avoided
Dooley et al. (2008)	–	1–14		1–14	15	6 cases, US$/t CO_2
McKinsey & Company (2008)						€/t CO_2 abated in 2020 and 2030, respectively
Early (after 2020)	25–32	4–6		4–12	35–50	
Mature (after 2030)	18–25	6–8		6–14	30–45	
Hildebrand (2009)	30–40		5–15		35–55	US$/t CO_2 captured
Eccles et al. (2009)	–	–		2–7	–	2005 US$/t CO_2

Numbers indicate cost per tonne of CO_2 captured unless otherwise indicated
Transport costs are for stand-alone cases unless otherwise indicated
IGCC: integrated gasification combined cycle: NGCC natural gas combined cycle: PC pulverized coal

Ploumen et al. (2007) provide an in-depth analysis of the capture costs, but their analysis does not include any assessment of transport and disposal costs. They conclude that new IGCC power plants are the most likely candidates for CO_2 capture at costs in the range of €25–30/t CO_2. This is considerably lower than PC plants, where the capture cost range is €32–42/t CO_2. The authors note, however, that IGCC is a less mature and more expensive technology than PC and this may involve additional costs to utilities. The capture costs in power plants retrofitted with capture equipment are estimated to be in the range of €40–52/t CO_2.

Hendriks et al. (2007) assess transport costs in the complex terrain of the Netherlands (densely populated areas, waterways, freeways). The average cost for 100 km is estimated at €1.6/t CO_2 and €1.0/t CO_2 for transporting 10 and 20 Mt CO_2/year, respectively, in large networks.

Pöyry Energy Consulting has developed a model to examine how the economics of the entire CCD chain might evolve in the UK with the increasing deployment of this technology (Pöyry Energy Consulting 2007). The Pöyry Energy Consulting model estimates the abatement costs of the three main stages of the CCD process, these being:

- CO_2 capture at the emission sources (large fossil-based power plants and industrial sites in the UK);
- CO_2 transport to the disposal site (including optimization of the transportation network);
- CO_2 disposal in offshore oil- or gasfields and aquifers.

The report concludes that using CO_2 for EOR can generate revenue up to £16/t CO_2, which partly compensates for other CCD costs (before any taxation issues are considered); the cost of disposing of CO_2 in aquifers is close to £1/t; and the cost of CO_2 disposal in oil- and gasfields ranges from £1/t to £20/t. The cost for abatement of up to 100 Mt CO_2/year is estimated to be in the range of £21–28/t.

The Pöyry Energy Consulting report (Pöyry Energy Consulting 2007) also includes an example with detailed cost calculations for a 485 MW coal-fired unit at Aberthaw B in 2015, from where the captured CO_2 is transported by pipe (over a distance of 370 km) to a gas terminal and then to an offshore aquifer (an additional 85 km) with ideal features (shallow water, shallow depth of the disposal media, large disposal capacity to support 46 wells from two platforms). Transport costs of the captured CO_2 from all three units of the Aberthaw B power plant to the disposal site are estimated in terms of CO_2 abated and amount to £4.53/t (£3.65/t CO_2 captured) while disposal costs coming to £1.21/t (£1.01/t CO_2 captured).

An IEA report (IEA 2008) presents a comprehensive technology analysis for CCD. The study includes an overview of the prospects and costs, the legal and regulatory frameworks as well as financing issues, together with status reports of CCD activities in many countries. The chapter on capture technologies also includes cost assessments presented as unit costs of CO_2 captured and avoided for coal- and gas-fuelled electricity generation (see Table 1). The cost estimates of transport and disposal technologies are less detailed and arrive at a cost of US$1–6/t CO_2 transported. The disposal costs in Europe are estimated in the range of US$10–20/t CO_2 in saline aquifers and US$10–25 US/t CO_2 in depleted oil- and gasfields. Cost estimates for North America are similar: US$15–25/t CO_2 in similar geological formations.

The report by The Boston Consulting Group (BCG 2008) confirms that CCD is a technically feasible solution. It briefly discusses examples of ongoing CCD activities in several countries and concludes that ‘Over the long term, the technology would pay for itself at a stable carbon price of €30/t’ (p. 4). The main cost components are estimated in the order of €25 to capture, €2–3 to transport and €4–5 to dispose of a tonne of CO_2. These proportions are assessed to be stable across the main global regions and in accordance with the assumed declining costs between 2008 and 2030.

Hamilton et al. (2008) present a financial analysis for new supercritical PC plants with CCD for the nth plant (i.e. involving some cost decline due to accumulating experience). The authors estimate the costs of *avoided* CO_2 emissions (always larger than the capture costs by a factor of 1.1–1.4) and focus on the cost of CO_2 capture—transport and disposal are considered at an estimated US$10/t CO_2 avoided. Their CCD estimate shows that post-combustion capture in a new supercritical coal plant would cost US$52/t CO_2 avoided, amounting to US$62/t CO_2 taking account of transportation and disposal costs of US$10/t CO_2. This avoidance cost is significantly higher than carbon price estimates resulting from various CO_2 regulation proposals in the USA; slightly above even the highest carbon price of US$61/t CO_2 estimated for the year 2030 under the Lieberman-Warner bill.

Looking beyond individual site- and project-specific conditions and costs, Dooley et al. (2008) estimate the long-term average price for CO_2 transport and disposal (including measurement, monitoring and verification) at US$15/t CO_2 (2005 US$). The authors present six actual modelled cost split cases across which both the transport and disposal costs vary between about US$2–14/t CO_2 (depending on the CO_2 source and flow rate, the transport distance and terrain and the features of the disposal site) but the sum of these two components is around US$15/t CO_2 in each case. Hence the authors argue that the US$15/t CO_2 is a robust estimate for transport and disposal costs, likely to prevail in many CCD projects in the USA.

The study by McKinsey and Company (2008) provides an overview of CCD in Europe in three phases, with the primary focus on the economic aspects: the demonstration phase up to 2015, the early commercial phase around and shortly after 2020 and the mature commercial phase commencing after 2030, if by then at least 80–120 projects are implemented in Europe to foster the learning effect. Along this path, CCD costs are estimated in the order of €30–45/t CO_2 *abated* by 2030 (see Table 2). In the reference cases, transport and disposal costs remain relatively stable across the various phases at €4–8 and €4–14/t CO_2, respectively.

The McKinsey report identifies capital costs (cost of CCD equipment per kW plant capacity) and the average cost of capital as the main uncertainty factors affecting the total costs of CCD. Construction and material costs for CO_2 pipelines are highly proportional to their length; hence, distance is the most sensitive factor in the transport cost. Yet, because of the low share of transport in the total costs, this effect is limited: doubling the transport cost would lead to only about a 10% increase in the total CCD cost.

Table 2 Costs of CCD in the deployment pathway in €/t CO_2 abated

	Assumptions	Capture	Transport	Disposal	Total[a]
Demonstration phase (2012–2015)	300 MW hard coal or lignite, 25 years lifetime, 80% utilization rate, 100 km onshore, 200 km offshore transport	51–64	5–7	5–13	60–90
Early commercial phase (after 2020)	900 MW hard coal or lignite, 40 years lifetime, 86% utilization rate, 200 km onshore, 300 km offshore transport, 1,500 m injection depth, onshore DOGF—offshore saline aquifers	25–32	4–6	4–12	35–50
Mature commercial phase (after 2030)	900 MW hard coal or lignite, 40 years lifetime, 86% utilization rate, 300 km onshore, 400 km offshore transport	18–25	6–8	6–14	30–45

Source: McKinsey & Company 2008
DOGF: depleted oil- and gasfields
[a] Totals rounded to nearest 5

Disposal costs largely depend on the location (onshore versus offshore) and the nature of the disposal site. The McKinsey report estimates that disposal in saline aquifers might cost 10–15% more than in depleted oil- and gasfields because of the limited amount of information available about the former; hence the need for more exploration and site characterization. (This indicates the massive dependence of the disposal cost on local conditions: cf. the Pöyry Energy Consulting (2007) model, in which disposal in aquifers is estimated to cost less than in oil- and gasfields.) The importance of this cost component is the basis for economies of scale: unit disposal costs at a large disposal site serving two commercial power plants could be 30–35% lower than at sites serving only one plant, whereas they might be 60–70% higher if two small sites are needed for disposing of the captured CO_2 from a single plant. Finally, the McKinsey report notes that EOR and EGR could considerably reduce the overall cost of CCD. However, these options increase the possible range of CO_2 disposal costs even further, making them the most variable cost factor in relative terms. Nonetheless, given the relatively small fraction of global CO_2 generation that could be used for EOR/EGR and the low share of the disposal component in the total CCD costs, the ultimate effect of this large cost variation is modest in global CCD cost accounting.

Hildebrand (2009) presents an in-depth assessment of capture costs by exploring the options of partial capture as opposed to full capture (capturing nominally 90% of CO_2), which involves significant penalties on the technology, plant performance and capture costs. She presents spreadsheet models for both PC and IGCC plant technologies and investigates plant performance and economics as a function of capture percentage. The results show that partial capture can preserve efficiency and that, for a PC plant, the cost savings associated with partial capture are significant. The costs of captured emissions are estimated to be in the range of US$30–40/t CO_2. These numbers do not include the costs for transportation, disposal and monitoring, which can add US$5–15/t.

Focusing on the disposal part of the CCD chain, Eccles et al. (2009) present a general model that represents the maximum CO_2 disposal potential, the maximum injection rate and the cost of CO_2 disposal. By applying the model to deep saline aquifers in sandstone reservoirs in the USA, the authors observe essential linkages between injection rates and the amount of CO_2 that can be stored (increasing with depth, hence decreasing unit costs) and the cost of drilling and injection equipment (increasing with depth, hence increasing unit costs). Characteristics of the reservoirs vary significantly even within the same basin and the actual disposal costs will diverge accordingly, between US$0.01 and US$100/t CO_2. What is more important, however, is that the model estimates US$2–7/t CO_2 for the full range of depth and basin properties for formations not deeper than 3,000 m with what the authors consider base case thickness (65 m) and permeability (22 mD). The authors conclude that in the USA regions with extremely low injections costs exist in many reservoirs but the total capacity of low-cost regions is 'likely to be much lower than the thousands of gigatons often cited as the potential storage capacity of deep saline aquifers' (Eccles et al. 2009, p. 1967).

There are many other issues beyond the direct cost figures that will influence the diffusion of CCD. Narita (2009) maintains that the absence of secondary benefits and uncertainties associated with CCD would require thorough cost-benefit comparisons with other CO_2 mitigation options to be conducted. The author frames his CCD assessment as utilization of a non-renewable resource with a limited capacity. In this framework, scarcity of geological disposal capacity should involve a shadow price which could raise the effective price according to a Hotelling rent. By using a simple analytical dynamic optimization model, Narita examines the optimal paths of CCD use by comparing the operational price with the real price, including the shadow price. He concludes that the inclusion of the shadow price of CCD could make the technology more expensive and thus relatively less attractive compared to, for example, renewable energy sources.

A report by the National Energy Technology Laboratory estimates the costs of CCD in terms of both capital cost and the long-term cost of electricity (NETL 2007). The study includes a detailed breakdown of the likely cost of CCD for the major types of fossil fuel-fired power plants. This report also found that retrofitting CO_2 capture to today's power plants using existing technology is expensive.

For PC plants, the cost of CO_2 capture, transport and disposal in an underground formation could add 70–100% to the cost of electricity. IGCC power plants can achieve a lower cost for CCD because, in this case, CO_2 can be captured from the gas stream from gasifying coal. Yet NETL (2007) estimates that adding present-day CCD technology to IGCC plants would increase the cost of electricity by at least 30%.

Groenenberg and de Coninck (2007) investigate a series of policy instruments for reducing greenhouse gas emissions in the EU. They conclude that while the European Union Emission Trading Scheme (EU ETS) is the most cost-effective instrument, it is difficult to project the level of incentives that it provides in the future for CCD activities. The incentives may remain too weak if the allocation continues to be based on National Allocation Plans, grandfathering and limited harmonization by the European Commission. In this case, the EU ETS is not likely to provide sufficient deployment of CCD in the short term, or even in the longer term because of its short time horizons and because of a lack of commitment by the EU to deep emission reductions over the long term.

Celebi and Graves (2009) observe that CO_2 mitigation policies using cap and trade schemes with a drastic near-term mitigation limit are likely to lead to highly volatile CO_2 prices. This volatility will significantly increase investment risks in mitigation projects like CCD, raise the cost of capital, and thus discourage investments. The authors estimate that CO_2 price volatility could delay investment in CO_2 mitigation technologies by 10 years or more. Their proposed solution to this problem is a safety valve mechanism that involves both a floor and a ceiling on CO_2 prices.

The small sampler of recent cost estimates presented in this section indicates large variations in all phases of the CCD chain. (Some variation in capture costs is due to the timing of the cited report, since costs have been increasing significantly over recent years (see the note about cost escalation above)). This is despite the possibility of drawing on the actual costs of many components observed in related industries, especially oil and gas drilling and transport. In relative terms the smallest variation is in capture costs followed by a somewhat larger variation in transport costs. Disposal costs tend to have the widest ranges in relative terms because of the large range of possible disposal formations and the possibility of revenue generation. However, as capture costs dominate by far the total CCD price tag, even the larger variation in disposal costs causes only a small variation in downstream costs and an even smaller one in the total electricity cost.

2.3 Relative Importance of the Disposal Costs

Considering the diversity of the CO_2 disposal options and the resulting wide range of disposal cost estimates, any assessment of the level of disposal cost per unit of electricity or of the relative importance of the disposal costs in the total fuel cycle costs and in the levelized cost of electricity (LCOE) should be handled with care. This section presents some rather broad calculations based on the disposal costs found in recent literature and presented in the previous section.

The estimated disposal costs per unit of electricity generated vary across a very wide range (see Table 3). The main driver of this variation is the targeted geological formation (saline aquifers versus depleted oil- and gasfields), while the CO_2 intensity of the power generation technologies plays only a minor role.

Combining the estimated CO_2 disposal costs with the fuel costs (taken from recent cost estimates of capture-relevant technologies) allows us to calculate the share of the former in the total fuel cycle costs. Cheap disposal options represent only a low share (a few per cent) in the total fuel cycle costs, while expensive geological targets can amount to 35–40% of the fuel cycle costs. Not surprisingly, this pattern can also be observed when we calculate the share of the disposal cost in the extended LCOE (base plus disposal costs). This portion represents a very low share for low-cost disposal options, but can reach 15–20% for expensive geological options.

It is important to emphasize that the numbers in Table 3 result from, at best, indicative conceptual calculations. The basic insights concerning the dominance of the geological formations as the main driver of the disposal cost are robust, but the numbers should not by any means be considered as precise estimates.

3 Costs of Radioactive Waste Disposal

3.1 Overview of the Main Cost Items

The International Atomic Energy Agency's (IAEA) new General Safety Guide on Classification of Radioactive Waste (Safety Standards Series No. GSG-1) (IAEA 2009a) classifies RW primarily on the basis of long-term safety considerations and the associated disposal options. High-level waste (HLW) is waste with radioactivity levels high enough to generate significant quantities of heat through the radioactive decay process or with large amounts of long-lived activity. Disposal in deep stable geological formations with engineered barriers is an option considered appropriate for the disposal of HLW. Intermediate level waste (ILW) is waste which, because of its content, requires a higher level of containment and isolation than is provided by near-surface disposal, however, with no or only limited provision for heat dissipation during its storage and disposal. A repository for ILW is distinguished from a repository for HLW by the degree of integrity and stability of the geological formation, and not necessarily by the depth of the repository, although the repository for ILW is sometimes referred to as an intermediate-depth disposal as opposed to deep geological disposal for HLW. Deep disposal of ILW has also been discussed, but mainly for social and economic rather than safety reasons.

For the purposes of comparing the cost of RW and CO_2 disposal, we focus on HLW, including SNF, for which the deep geological repository concept is generally envisaged on the grounds of long-term safety considerations.

Cost studies for the following waste repositories are the main sources of the data discussed in this section: a final waste repository (which has now been suspended) at Yucca Mountain in the USA (OCRWM 2008a), a final waste repository facility at Olkiluoto in Finland (Kukkola and Saanio 2005), a final waste repository at

Table 3 The share of CO_2 disposal costs in the fuel cycle and levelized electricity costs

	Disposal cost (US$/t CO_2)	Emissions (t CO_2/MWh)	Disposal cost (US$/MWh)	Fuel costs (US$/MWh)	F+D costs (US$/MWh)	Disposal cost share in F+D (%)	LCOE (US$/MWh)	LCOE+disposal cost (US$/MWh)	Disposal cost share in LCOE (%)
McCoy and Rubin (2005) – PC	1.63	1.07	1.74	19.78	21.52	8.08	88.64	90.38	1.92
McCoy and Rubin (2005) – IGCC	1.63	0.9	1.46	19.81	21.27	6.88	93.44	94.91	1.54
Pöyry Energy Consulting (2007) aquifers	1.68	0.81	1.36	29.06	30.42	4.46	100.90	102.26	1.33
Pöyry Energy Consulting (2007) oil/gas fields	33.53	0.81	27.16	29.06	56.22	48.31	100.90	128.06	21.21
Vosbeek and Warmenhoven (2007)	3.81	0.77	2.94	29.01	31.94	9.19	91.88	94.82	3.10
Dooley et al. (2008) – LDC	1.03	1.02	1.05	20.99	22.04	4.77	91.11	92.16	1.14
Dooley et al. (2008) – HDC	14.44	1.02	14.73	20.99	35.71	41.23	91.11	105.84	13.91

McKinsey & Company (2008) LDC, LE	11.18	0.90	10.06	29.06	39.12	25.72	90.81	100.87	9.98
McKinsey & Company (2008) LDC, HE	11.18	1.00	11.18	29.06	40.24	27.79	90.81	101.99	10.96
McKinsey & Company (2008) HDC, LE	26.09	0.90	23.48	29.06	52.54	44.69	90.81	114.29	20.55
McKinsey and Company (2008) HDC, HE	26.09	1.00	26.09	29.06	55.15	47.31	90.81	116.90	22.32
Eccles et al. (2009) LDC	2.02	1.02	2.06	20.99	23.05	8.93	91.11	93.17	2.21
Eccles et al. (2009) HDC	7.06	1.02	7.20	20.99	28.19	25.55	91.11	98.32	7.33

Disposal costs: based on Table 2, converted to 2010 US$ by using currency-related GDP deflators and 2010 exchange rates
Specific emissions: taken from the source study if included or industry average from the IEA or US DOE online databases
Fuel costs: based on fuel cost estimates in IEA and NEA (International Energy Agency and Nuclear Energy Agency) (2010)
LCOE (levelized cost of electricity): based on IEA and NEA (2010) at 10% interest rate
F + D costs: fuel cost + disposal cost (without capture and transport): LDC low disposal cost: HDC high disposal cost: LE low CO_2 emissions/MWh: HE high CO_2 emissions/MWh

Forsmark in Sweden (SKB 2003), an unidentified final waste repository in Belgium (ONDRAF/NIRAS 2001a), an unidentified final waste repository in Japan (METI 2008a), an unidentified regional joint waste repository of 14 EU countries (Chapman et al. 2008) and an unidentified final waste repository in the UK (Nirex 2005). These studies vary in terms of their coverage, assumptions, level of detail and uncertainty, and transparency of the cost estimation methodology. The costs quoted are mainly overnight costs (i.e. interest to be accrued during construction and price escalation effects is ignored and future costs streams are not discounted, unless otherwise noted).

Generally speaking, the cost estimates for the radiological waste disposal consist of the following elements in the three main phases:

- *Pre-operation phase*: site investigation and characterization, and development and construction of a repository, transportation system to a repository, and encapsulation plants and other above-ground facilities;
- *Operation phase*: transportation, encapsulation and emplacement of wastes;
- *Post-operation phase*: decommissioning of the above-ground facilities and closure and monitoring of the repository.

Throughout all the phases, costs for programme administration are incurred. R&D may or may not be explicitly included. When SNFs are reprocessed, reprocessing costs and disposal costs of associated long-lived low and intermediate level wastes (LILW-LL) are not included. Table 4 summarizes what the total costs include in the above-mentioned studies. To the extent possible, the terminology used in the respective original studies is used when the cost components are being presented. Details of each study and definitions of some of the terminology are given in Sect. 3.2.1.

During the *pre-operation phase*, activities accounted for in the cost estimates for site investigation and characterization may include land acquisition costs (Japan and Belgium), costs related to site selection and conceptual designs for the development of the repository (USA). This process may take 10–30 years, including a period for preliminary siting studies and licence approval.

Construction of the underground facility (repository) and the above-ground facilities (encapsulation plants, on-site/off-site infrastructure, administration buildings, etc.) are also major cost items for the pre-operational phase. It may also include costs related to licensing, design, management, engineering and procurement. Construction of the underground facility may take 5–10 years. Typically, the operation and construction are planned to go partially in parallel, and for that reason not all construction costs occur during the pre-operational phase. For example, in the UK estimates, the construction costs after the first waste emplacement account for 32% of the total repository construction costs, mainly due to the construction and fit-out of the remaining disposal tunnels.

Some estimates include the costs for a waste transportation system. In the USA, the waste transport was planned to be handled mostly by rail, using dedicated trains. Costs for acquiring rail, truck cask systems and rolling stock for the national transportation system, as well as the costs for providing the interface between the

Table 4 Cost components and estimates for radioactive waste disposal in selected studies[a]

Country	Site investigation/ characterization	Underground facility	Above-ground facility	Post closure monitoring	Transportation	Programme administration	Other	Total
USA 2007 US$ (million)	Repository development and evaluation costs US$8,330 M	Licensing costs, engineering, procurement and construction cost, emplacement operation costs, pre-closure monitoring costs and closure costs US$56,400 M (4% for licensing, 28% for other engineering, procurement and construction, 47% for emplacement operations, 18% for monitoring, and 2% for closure)			Transportation costs US$20,250 M	Balance of programme costs US$11,200 M (21% for development and evaluation, 7% for quality assurance, 3% for waste management, 29% for programme management, 28% for benefit paid to the local entities, 12% for funding other agencies)	–	US$96,180 M
Finland December 2003 € (million)	Not included	Repository costs €714 M (investment 50%, operation 34%, closure 15%)	Above-ground facility (encapsulation plant, operating building, ventilation shaft building, and research building) €1,827 M (investment 8%, operation 92%, decommissioning less than 1%)	Not included	Accounted as a part of operational costs of above-ground facilities (€0.3 M)[b]	–	Contingency of 20% is included in each cost item	€2,542 M

(continued)

Table 4 (continued)

Country	Site investigation/ characterization	Underground facility	Above-ground facility	Post closure monitoring	Transportation	Programme administration	Other	Total
Sweden January 2003 SEK (million)[c]	Deep repository—siting and site investigation SEK 2,058 M	Deep repository—spent fuel SEK 8,150 M (investment 56%, operation and maintenance 14%, decommissioning and backfilling 30%)	Encapsulation plant, deep repository off-site facilities, deep repository above-ground facilities SEK 13,783 M (investment 32%, operation and maintenance 66%, decommissioning 2%)	Not included	Transport (for all waste types) SEK 3,024 M (investment 52%, operation and maintenance 48%)[d]	RD&D and administration costs SEK 9,692 M	Incurred costs are included under each cost categories	SEK 36,707 M
Japan 2008 ¥ (billion)	Site investigation and selection and land acquisition ¥174 B	Design and construction, operation, monitoring, and decommission and closure ¥1,823 B (design and construction 50%, operation 40%, monitoring 5%, decommissioning and closure 6%)		Included implicitly in the project administration costs		Project administration ¥553 B	Value-added tax ¥103 B Technology development ¥104 B	¥2,757 B
Belgium 2000 € (million)	Land acquisition, site characterization and site preparation	Building licence, off-site infrastructure, surface facilities, excavation, equipment for post-conditioning, handling equipment, backfilling 0of galleries, operating costs and personnel costs					Neither incurred R&D expenditure of €150 M nor additional R&D expenditure of €75–300 M is included in the total costs	

Reprocessing option	€23.4 M	€554.8 M (63% for investment phase, 22% for waste emplacement phase, 15% for closure phase)[e]		Not included	Not included	Not explicitly reported	50% contingency on average is included under each cost item	€578 M
Direct disposal option	€41.4 M	€1452.4 M (68% for investment phase, 10% for waste emplacement phase, 22% for closure phase)		Not included	Not included	Not explicitly reported	61% contingency on average is included under each cost item	€1,494 M
UK June 2004 £ (million)	Site characterization and rock characterization facility				Transport costs	Nirex internal and other programme works		
Non-retrievable option[f]	£755 M	£1,735 M (46% for construction, 47% for operation, 7% for sealing and closure)	£928 M (41% for construction, 57% for operation, 1% for sealing and closure)	Not included	£302 M[g]	£960 M		£5,226 M
Retrievable option		Common to above-ground and underground facilities £546 M (73% for construction, 27% for sealing and closure)		Maintenance and refurbishment for 250 years £1207 M[g]				£6,765 M
European Commission 2006 € (million)	Repository siting	Repository construction, operation and closure	Encapsulation facility		Transportation	Repository related R&D and administration		

(continued)

Table 4 (continued)

Country	Site investigation/ characterization	Underground facility	Above-ground facility	Post closure monitoring	Transportation	Programme administration	Other	Total
Swedish Cost Model	€231 M	€4,121 M (45% for construction, 34% for operation, 21% for closure)	€2,633 M	Not included	€1,065 M	€1,091 M		€9,141 M
Swiss Cost Model	€512 M	€4,698 M (45% for construction, 28% for operation, 27% for closure)	€2,433 M	Not included		€320 M (including compensation)		€9,029 M
Finnish Cost Model	Not explicitly reported	Repository facilities, operations and closure €2,514 M (54% for facilities, 40% for operations, less than 6% for decommissioning)	Above-ground facilities, operations and decommissioning €7,083 M (2% for facilities, 98% for operations, less than 1% for decommissioning)	Not included		Not included		€10,662 M

[a] The shares may not sum up to 100% due to rounding

[b] Already included in the cost of the above-ground facility

[c] The incurred costs through 2003 are given in current prices, and future costs estimates after January 2004 are given in the constant prices of January 2003. In this presentation, inconsistencies of the reported prices areignored and a simple sum of incurred costs and future costs is presented

[d] This share is applicable only to the future costs, as the share for the incurred costs are not reported

[e] The Belgium report presents the costs according to the phases

[f] Non-retrievable costs do not include maintenance and refurbishment costs

[g] This estimation is taken from a separate study (Nirex 2006), in which the price level of September 2003 was used. It has been adjusted to the price level of June 2004 using a conversion rate of 1.006

national transportation system and the repository, are included in their cost estimation. In Finland, the SNF will be transported by road. The transportation costs include transporting trailers and SNF transport casks. The SAPIERR II project (details below) also provides a rough estimate based on the weight of waste to be handled, without considering the transportation distances.

R&D costs may or may not be included explicitly in the total cost estimates. In the case of Japan, a cost item called 'technology development' is included. In the case of Finland and Belgium, R&D costs are explicitly excluded.

For the *operation phase* of a project, the costs largely depend on the amount of waste to be disposed of. Execution of waste transport, waste handling, purchase/manufacturing of the casks/canisters (USA and UK), buffer material production (Japan, UK, Finland), encapsulation and emplacement of waste packages into a repository are activities often accounted for in the total cost estimates.

During the *post-operation* phase of RW disposal, the main activities include decommissioning of the above-ground facilities, restoration of the surface area and closure and monitoring of the underground facility. Monitoring may include pre-closure and post-closure monitoring. In the case of the USA, permanently installed sensors would monitor waste packages, emplacement drift and the surrounding rock, providing the data to confirm performance during the pre-closure monitoring period of 50 years. In the US estimate, fabrication of drip shields which would be emplaced during this period constitutes a major part of the pre-closure monitoring costs. Closure activities include backfilling of shafts and ramps, sealing, and protection of the repository from unauthorized intrusion. The period assumed in the cost estimates for closure cover 3–20 years. The UK estimates involve longer periods for post-closure monitoring, foreseeing up to 300 years.

Administration costs may include safeguards and security activities, regulatory, infrastructure and management support costs. Other miscellaneous costs included in the cost estimates are benefits paid to state and local entities (governments and tribes) (USA), contingency (Finland, Belgium) and value added tax (Japan).

3.2 Costs of Deep Geological Disposal of High-Level Waste

3.2.1 Cost Estimates from Various Countries

In the USA, Congress passed and the President signed a public law which approved Yucca Mountain as the site for a waste repository in 2002. The US Government announced suspension of all activities in 2009 (and the final decision is still pending), but the cost studies still provide valuable information. The Nuclear Waste Policy Act (NWPA) of 1982 put a limit on the emplacement of the SNF to 70,000 tonnes of heavy metal (tHM) in the first repository. Of the 70,000 tHM, 63,000 tHM is allocated to civilian waste. However, more than 58,000 tHM commercial SNF is already in storage, and the total inventory of commercial SNF is expected to grow at a rate of about 2,000 tHM/year. In 2008 the Secretary of Energy submitted, in accordance with the 1982 NWPA, a recommendation to the President and Congress that the current 70,000 tHM statutory limit should be removed, otherwise a second repository would be needed (OCRWM 2008b). Although the operation was expected to start no sooner than 2020, the programme has been at a standstill since February 2009 (WNN 2009).

Costs of disposal of the SNF and vitrified HLW are estimated by the Office of Civilian Radioactive Waste Management (OCRWM), providing a basis for assessing the adequacy of the Nuclear Waste Fund Fee as required by the NWPA (OCRWM 2008a). The latest estimates assume that all currently projected SNF and vitrified HLW from civilian and defence use will be disposed of at the Yucca Mountain repository. The projected amount referenced in the cost estimation is 122,100 tHM of SNF and vitrified HLW. This estimate is based on the discharge projections from all reactors operating until the end of licensed lifetimes, taking into account 47 reactors to which licence extensions were granted by the National Regulatory Commission (NRC) as of January 2007.

The total system life cycle costs span the period from 1983 to the assumed closure date of 2133, and total US$96.18 billion at a constant price of 2007. The total costs consist of costs for the repository, transportation and the balance of the programme, accounting for US$64.7 billion, US$20.3 billion, and US$11.2 billion, respectively. Of the repository costs, US$9.9 billion had been disbursed as of 2006, of which the 'development and evaluation' phase accounts for the major part (US$8.3 billion). The remaining repository costs are accounted for by the 'engineering, procurement, and construction' phase (32%, nearly half of which had already been disbursed), the 'emplacement operation' phase (the largest cost item, accounting for 47%, nearly half of which is due to the fabrication of waste packages), the 'monitoring' phase (18%, mostly due to fabrication of drip shields), and the 'closure' phase (2%). Part of the costs of the 'engineering, procurement, and construction' phase are accounted for by the licensing costs (4% points out of 35%). 'Monitoring' phase costs refer to costs related to pre-closure monitoring, which is assumed to last for 50 years after the emplacement operations.

Transportation costs consist of those related to the design of the transportation system, the National Transportation Project (transport from waste generating sources to the state of Nevada) and the Nevada Rail infrastructure project (providing an interface between the nationwide transportation system and the repository).

In Finland the Parliament ratified a decision-in-principle in 2000 for the construction of a final disposal facility for SNF at Olkiluoto (Finnish Ministry of Employment and the Economy 2001). The disposal facility will be constructed after the licence from Government is received during the period 2013–2019. The disposal facility will start operating in 2020, and continue its operation for over 100 years (Kukkola and Saanio 2005). Licences to construct and operate the final disposal facility are currently under development by Posiva, a company responsible for the final disposal of SNF. It is planned to dispose of the SNF generated from four existing and one new nuclear power plant (NPP) in the repository; the amount of SNF is estimated to be equivalent to approximately 6,500 tonnes of uranium (tU) (Finnish Ministry of Employment and the Economy 2002). Another application for a decision-in-principle on an extension to the final disposal facility for disposing of SNF from the second new NPP (Olkiluoto 4) was submitted to the Government in 2008 for approval (Posiva 2008a). If it is approved, the total amount of SNF to be disposed of increases to 9,000 tU.

The costs of the disposal facility at Olkiluoto were estimated by Posiva in 2005 (Kukkola and Saanio 2005). The estimates are based on the disposal of SNF corresponding to 5,643 tU. The total costs are estimated at €2,542 million, at a constant December 2003 price. They include transportation costs and contingency of 20% and do not include R&D costs and site selection costs. SNF will be transported to the encapsulation plant by road. The costs are divided into three periods, namely, pre-construction/construction, operation and closure, and account for 11%, 85% and 5% of the total costs, respectively. Across all these periods, investment-related costs account for 21%, whereas operation-related costs account for the rest. The main cost components are the operation costs of above-ground facilities (66% of total costs), of which 80% is accounted for by costs for encapsulation materials and personnel. Transportation costs are insignificant. These estimates are based on the reference design, in which the canisters with SNF are emplaced vertically in individual deposition holes. A separate study conducted by Posiva and SKB (Posiva 2008b) evaluated costs for an alternative design, KBS-3 H, in which the canisters are serially emplaced in long horizontal drifts. It was estimated that for the Olkiluoto site, KBS-3 H would realize savings of €96 million (at a constant price for an unspecified time period).

In Sweden site investigations began in 2002 at Forsmark and Laxemar. In June 2009 Forsmark was selected as a site for the final repository based on the results of these investigations. The repository is expected to have a capacity of 6,000 canisters (about 12,000 of SNF) and be located at a depth of about 500 m. The operation is expected to start by the beginning of the 2020s and to continue for about 40 years (SKB 2010). The costs for management and disposal of all kinds of RW were estimated in 2003 (SKB 2003). They cover costs for RD&D, transportation, a central interim storage facility for SNF, encapsulation of SNF, a deep repository for SNF, final repositories for LILW-LL, reactor waste, short-lived waste and waste produced during decommissioning, as well as costs of decommissioning the NPPs. The attribution of costs specific to a deep geological disposal of SNF is not provided in the study. However, it is fair to assume that the majority of the RD&D costs, some transportation costs, and all costs related to encapsulation and a deep repository for SNF are considered as costs associated with deep geological disposal of SNF.

The number of canisters referenced in the cost estimate is 4,500, corresponding to the existing and expected SNF of 9,493 tU. Of the total waste management costs for all waste categories, namely (Swedish krona) SEK 49,600 million, the subtotal related specifically to disposal of SNF was estimated to be SEK 29,870 million (approximately €3.2 billion) at a constant 2003 price. It includes SEK 4,860 million for RD&D and administration (including costs attributable to waste disposal other than SNF), SEK 2,230 million for investment, operation and maintenance of transport (including costs attributable to transportation of waste other than SNF), SEK 7,920 million for investment, operation, maintenance and decommissioning of an encapsulation plant (including canister plants), and SEK 14,860 million for the deep repository. The costs for the deep repository include the following cost categories: siting, off-site facilities (investment and operation), above-ground facilities (investment, operation, maintenance and decommissioning) and underground facilities (investment, operation, maintenance, decommissioning and backfilling), accounting for 7%, 2%, 36% and 55%, respectively, of total deep repository costs.

The above does not include incurred costs through 2003, which, at current prices, are estimated to be SEK 4,832 million for RD&D and administration (including costs attributable to waste disposal other than SNF), SEK 794 million for transportation (including costs attributable to waste disposal other than SNF), SEK 192 million for an encapsulation plant, and SEK 1,018 million for siting and site investigations for the deep repository, totalling SEK 6,837 million. For presentation in Table 4, the incurred costs are added to the projected costs for each cost item, making the total costs SEK 36,707 million.

In Japan, under the Specified Radioactive Waste Final Disposal Act adopted in 2000, the Basic Policy on disposal of vitrified HLW was established in 2000 and revised in 2008. The policy sets out a timeline, starting from site selection (to be completed by around 2028), to construction of a final repository facility and operation of the repository (starting from around 2033–2037). The minimum capacity of the repository should be 40,000 canisters, which are estimated to be generated from reprocessing of SNF from nuclear electricity generation by 2021 (METI 2008b). The basis of the calculation is that 1 GW/year of NPP operation produces 30 units of vitrified HLW (METI 2008c; NUMO 2004). This indicates a reference energy production of 11,670 TWh.

The final disposal costs were estimated by the Advisory Committee for Natural Resources and Energy for two rock types (soft rock and hard rock) (METI 2008a). They are not substantially different (costs for the soft rock type are 5% higher than the hard). The average of the total costs for the two rock types is (Japanese yen) ¥2,757 billion (approximately US$27 billion) at a constant 2008 price. Costs are given according to the following cost categories: technology development (8.5%), site investigation and land acquisition (12.1%), design and construction (20.8%), operation (21.8%), monitoring (9.7%), decommissioning and closure (1.3%), project administration (22.0%) and value added tax (3.7%). The expenditure for decommissioning and closure is assumed to be due between 2075 and 2099. Project administration costs are assumed to be due between 2100 and 2395, presumably for post-closure monitoring purposes. Underground and above-ground facilities considered in the cost estimates include off-site infrastructure (harbour facilities and dedicated roads), an encapsulation plant and buffer material production facilities. Information on the way that the costs are attributed to each of these facilities is not provided. Costs related to two underground research laboratories do not appear to be included.

In Belgium deep disposal of HLW is considered as the reference solution. Research will continue for several years before a concrete decision is taken on the way the waste will actually be disposed of and where (ONDRAF/NIRAS 2009). The

Belgian Agency for Radioactive Waste and Enriched Fissile Materials (ONDRAF/NIRAS) published a report in 2001 (ONDRAF/NIRAS 2001a, b) that includes a cost estimate for a deep disposal facility. A reference site used for cost estimating purposes is the Boom Clay beneath the Mol-Dessel nuclear zone. The reference design for the vitrified HLW shows that there will be a total of 3,915 waste packages (corresponding to 4,860 tHM of conventional uranium fuel, all reprocessed) to be disposed of. The design also assumes that it will not only be used for the HLW but also for LILW-LL, although the cost assessments relate solely to disposal of vitrified HLW and SNF. According to the reference timetable, detailed design and safety studies take 10 years, and construction, operation and closure altogether take 30 years (including 20 years attributable to LILW-LL), assuming they are carried out partially in parallel. The total cost (attributable only to disposal of vitrified HLW and SNF), including construction, operation and closure of the repository, as well as the contingency margins for each of them, is estimated at €578 million (of which 50% is accounted for by the contingency margin) at a constant 2000 price. The costs are divided into three implementation stages: construction, operation and closure, each accounting for 64%, 21% and 15%, respectively, of total costs. The contingency margins for the construction and operation stages are 95%, and for the closure phase 138%. The cost estimates do not account for the R&D costs, which were approximately €150 million for the period 1974–2000 (at a constant 2000 price). ONDRAF/NIRAS estimated that additional R&D spending of €75–100 million should be enough to enter a pre-project phase, which is site specific, assuming that disposal and the Boom Clay are confirmed, respectively, as the long-term management option and the host formation. Should the authorities indicate their preference for another geological formation, then R&D spending to attain the same objective would be €250–300 million.

The report also estimates the costs of a direct disposal option, in which reprocessing is assumed to stop after the reprocessing of the 630 tHM SNF foreseen under the existing contract, the remaining SNF being disposed of without reprocessing. In this case, construction, operation and closure are assumed to take 40 years (of which 22 years are specifically for deposition of the vitrified wastes and SNF). The cost would then be estimated at €1,494 million (61% being accounted for by a contingency margin) at a constant 2000 price. The shares for three implementation stages are 70% for construction, 9% for operation and 21% for closure. The respective contingency margins for each step are 140%, 170% and 200%. The costs are much higher with this option, even discounting the fact that much higher contingency margins are assumed. It primarily reflects the fact that the galley space required for the *disposal* of vitrified HLW and SNF is much larger than is the case with a reprocessing option (a total length of 44 km instead of 6.5 km) and that the total length of the *main* gallery, to which the disposal galleries are connected, needs to be longer (4,245 m rather than 760 m) to allow the increased gallery spacing required for the SNF disposal.

In the UK, Nirex (2005) estimated costs for a repository for vitrified HLW and SNF, based on the Swedish repository concept, KBS-3. Cost estimates are based on the assumption that the repository would be a stand alone facility for HLW/SNF.

A reference timetable assumed in the cost estimation is: site characterization from 2007 to 2020, construction and underground research from 2020 to 2040, operation of the facility from 2040 to 2090, and closure from 2090 to 2100. The total number of canisters to be disposed of is 7,088 units, of which 3,700 units are for vitrified HLW, 572 units for SNF from pressurized water reactors (PWRs), and 2,816 units for SNF from advanced gas-cooled reactors (AGRs). To what extent these units correspond to weight is not stated. Although there is no direct reference to it, the UK Committee on Radioactive Waste Management (CoRWM) provides the national baseline inventory of RW in the UK (CoRWM 2005). Assuming conservatively that reprocessing of SNF will be discontinued (although reprocessing of all the existing and future SNF is planned), it consists of 54,500 tU of existing reprocessed SNF, which corresponds to 1,290 m^3 of HLW, 1,200 tU of PWR SNF, 3,500 tU of AGR SNF and 125 of plutonium and highly enriched uranium.

Total costs are estimated at £4.9 billion (approximately US$9 billion) at a constant June 2004 price. This total cost does not include transportation costs, post-closure costs and contingencies. The total costs of £4.9 billion are broken down into 'site characterisation', 'rock characterisation facility', 'repository construction to first waste emplacement', 'repository construction post first waste emplacement', 'repository operation', 'repository sealing and closure', and 'Nirex international and other programme works'. Repository construction is the major cost category (32%), followed by operation (27%) and programme works (19%). Above-ground facilities include a canister factory, an encapsulation plant, a bentonite/backfill plant, off-site infrastructure and other on-site infrastructure.

There is no explicit mention as to whether disposal of plutonium and enriched uranium is included in the cost estimates. In a similar study conducted by Nirex (2006), costs for disposal of plutonium (mainly from civilian sources) and enriched uranium (mainly from military sources) are estimated at £1.6 billion at a constant September 2003 price, on top of the £5 billion estimated for disposal of vitrified HLW and SNF. Furthermore, the operation period would be extended by 15 years. The transportation costs of canisters from waste generating sites to the repository were also estimated in this study but turned out to be minor (£0.3 billion). The total costs presented in Table 4 include the transportation costs. Should retrievability be retained as an option, the costs for maintenance and refurbishment before complete sealing of the repository, estimated at £1,207 million, should be added. It is assumed that maintenance and refurbishment will take place between the 50th year (at the end of the emplacement phase) and the 300th year from the first waste emplacement.

The SAPIERR II (Strategic Action Plan for Implementation of European Regional Repositories) project, supported by the European Commission and with the participation of 14 European countries (Austria, Belgium, Bulgaria, Croatia, Czech Republic, Hungary, Italy, Latvia, Lithuania, the Netherlands, Romania, Slovakia, Slovenia, Switzerland), published a report that includes cost estimates for a multi-national common repository programme (Chapman et al. 2008). Cost estimates were based on the waste inventory data and time schedule established in the predecessor project, SAPIERR (Support Action: Pilot Initiative for European

Regional Repositories) (Štefula 2006). The volume of non-processed SNF stored in SAPIERR countries by 2040 was estimated to amount to 25,637 tHM, based on the assumptions that (1) no new nuclear power reactors will be built, (2) the existing ones will operate until the end of their operational life time, and (3) there will be no plant life extension. The volume of vitrified HLW from SNF reprocessing by 2040 is estimated at 355 m^3, which roughly corresponds to 3,220 tHM of SNF, with the remaining volume of SNF being disposed of as SNF without reprocessing. (Note that for Bulgaria, Italy and the Netherlands, the volume of HLW was given only in terms of mass (150 m^3)). A conversion rate of 9 tHM/m^3, obtained based on Belgian and Swiss inventory reports (Štefula 2004), was used to derive the SNF equivalent of 1,350 tHM. The reference time schedule for a repository is start of repository operation in 2035, with the total length of operation being 50–60 years.

The cost estimates were prepared for six scenarios, four of which assume the joint disposal of HLW and ILW, and two of which assume repositories for disposing exclusively of SNF and vitrified HLW. Two scenarios correspond to the different rock types (hard rock and sediment rock). For the hard rock type, €8.1 billion (using a Swedish cost model) and €9.6 billion (using a Finnish cost model), both at a constant 2006 price, were estimated as costs for a repository and an encapsulation plant. For the sedimentary rock, €8 billion at a constant 2006 price was estimated (using a Swiss model). The costs for an encapsulation plant and for a repository were distinguished under the Swedish and Swiss cost models, and the costs for an encapsulation plant account for slightly over 40% in both cases whereas costs for the repository account for the rest. Three cost models were applied assuming that the disposed waste would be the half the reference volume. In this case the costs were estimated at €4.7–5.2 billion, indicating economies of scale effects. Indicative transportation costs were estimated at up to €1 billion, assuming unit costs for SNF transport were €40,000/t for the international transports that a European regional repository would require. The mode of transport is not specified and the estimate is based solely on mass. The total cost presented in Table 4 includes this transportation costs.

3.2.2 Amount of Radioactive Waste from Nuclear Power Generation and the Disposal Capacity

Most HLW arises as SNF from the operation of NPPs and as vitrified HLW from reprocessing of SNF. The amount of waste arising is determined mainly by the amount of electricity produced and the choice between direct disposal or reprocessing of SNF. The amount of waste generated is then used as a key parameter in estimating disposal costs. Assumptions regarding electricity production and the extent to which reprocessing of SNF is applied are used in the cost estimates discussed in the previous section, and are summarized in Table 5. Among the reports reviewed, only the Swedish report mentions explicitly the corresponding electricity generation. The value for Japan was calculated by the authors using the published ratio between the electricity production and the amount of vitrified HLW. Note that in the reviewed reports, the amounts of SNF are reported in different units. In subsequent sections, we present the amount of RW in terms of tHM (post-irradiation weight). In doing so, it was assumed that the unit quoted as tHM in various reports refers to the post-irradiation weight, rather than pre-irradiation weight, and that the unit quoted as tU refers to the initial weight of uranium in a UO_2 fuel assembly before irradiation. A ratio of conversion from an initial 1 tU of fresh fuel into fission products is applied to obtain the heavy metal weight of irradiated fuel. The conversion ratio is proportional to the burnup ratio (i.e. for a burnup ratio of 10 GWd/tU, the conversion ratio is 0.0105). In other words, for each 10 GWd/tU of burnup, the initial 1 tU becomes 0.9895 tHM, with the remaining 0.0105 having been converted into fission products.

There is a general relationship between the electricity generated and the weight of heavy metal in fresh fuel (the same as the weight of uranium in fresh fuel in the case of UOX fuel): the weight of SNF (tU) is approximated to be equal to electrical energy (GW/year) divided by the product of the efficiency (in per cent) and burnup ratio (GWd/tU). According to the authors' calculation using data from the IAEA's PRIS database (IAEA 2010a), the average net thermal efficiency and the burnup ratio of all power plants in the world including those shut down, weighted by the cumulative net electricity generated, are 32.9% and 35.7 GWd/tU, respectively. The average amount of SNF generated per GW/year of net electricity produced by all the reactors, weighted by the total cumulative net electricity production and converted into the weight of heavy metal using the above mentioned conversion procedure, is 39.9 tHM, while the averages for the PWR, boiling water reactor (BWR) and pressurized heavy water reactor (PHWR) subsets are 37.0, 41.8, and 165.1 tHM, respectively. In assessing how much SNF will be produced in the future, one should take into account that the amount of waste generated per unit of energy produced has been continuously reduced because of technological advances. For example, if we compute the average volume of SNF per GW/year excluding NPPs already shut down, then the amount of SNF per GW/year electricity is reduced to 38.5 tHM (with the PWR producing 29.2 tHM, the BWR 27.5 tHM and the PHWR 157.6 tHM). Note that in this calculation we assume that the weight of uranium in fresh fuel and the weight of heavy metal in fresh fuel is the same (i.e. UOX fuel is used). A fraction of this SNF is sent for reprocessing, producing HLW which is vitrified and stored for eventual final disposal. The remaining part of the SNF is also stored for eventual final disposal. As discussed in connection with the Belgium cost estimates, as the vitrified HLW requires about ten times less repository space than the equivalent amount of SNF, reprocessing reduces the overall space requirements of a repository for vitrified HLW and SNF, and thus decreases the costs. However, at the same time, reprocessing generates low-level waste (LLW) and ILW, which obviously increases the total costs by the amount required for their disposal. It is beyond the scope of this chapter to assess this trade-off.

Deep geological disposal of RW relies heavily on engineered barriers in addition to natural barriers. Construction of an underground facility requires massive underground engineering, which in turn implies some flexibility with respect to

Table 5 Assumptions on the amount of the wastes to be disposed of used in the respective cost estimates in Table 4

Study	Reference energy production	Spent fuel generated	Portion of spent fuel reprocessed	Expected number of waste packages to be disposed of
USA	Until the end of service life of respective NPP	122,100 tHM (of which 109,300 is commercial SNF)	10,300 tHM (belonging to US DOE with unknown share of commercial origin)	17,450 units of waste packaging
Finland	Until the end of service life of respective NPP	5,643 tU	None	2,899 units of canisters
Sweden	2,769 TWh	9,493 tU (all is due to commercial electricity production)	197 tU (producing no HLW)[a]	4,500 units of canisters
Japan	11,670 TWh until 2021	NA	All spent fuel is reprocessed	40,000 units of canisters
Belgium *Reprocessing option*	Until the end of 40 years of service life	4,860 tU (UOX fuel) and 70 tHM (MOX fuel)	4,860 tU of conventional uranium fuel	3,915 units of vitrified HLW, 144 units of MOX fuel
Direct disposal option			630 tU of conventional uranium fuel	420 units of vitrified HLW, 9,715 units of uranium spent fuel, 144 units of MOX fuel
UK	n.a.	n.a. Base line inventory: 59,200 tU	n.a. Base line inventory: 54,500 tU	3,700 units of HLW canisters, 3,388 units of SNF canisters
SAPIERR (EC)	Until the end of service life of respective NPP	25,637 tHM excluding those reprocessed	355 m^3 [b]	13,246 units of SNF canisters, 2,021 units of HLW canisters

NPP: nuclear power plant, *SNF*: spent nuclear fuel, t*HM*: tonnes of heavy metal, *tU*: tonnes of uranium, *US DOE*: United States Department of Energy

[a]This is because that reprocessing takes place abroad

[b]According to the author's calculation, this roughly corresponds to 3220 tHM of spent fuel

the capacity, compared to CCD, for which capacity is primarily defined by the availability of a suitable geological formation at a given site.

The density of HLW/SNF disposal in a disposal gallery is determined by thermal conditions, such as decay heat, and properties in the buffer and in the surrounding rock, as well as the requirement to ensure that the possibility of criticality will not be a concern. Greater thermal loads can be accommodated by extending the time that the repository is open and ventilated prior to repository closure. How the wastes are loaded in waste packages and whether the waste packages are stored to allow decay prior to emplacement are also key parameters determining the amount of waste that can be placed in a given volume of rock (OCRWM 2008b).

A repository for HLW is typically designed in such a way that all the expected waste to be dealt with in a given jurisdiction is disposed of at a selected site, and the capacity of a repository is thus determined primarily by the amount of expected waste in the foreseeable future. As discussed later, there is a strong economies of scale issue. Extending capacity at a later stage may be possible with a relatively small marginal increase in costs, as fixed cost components may account for a significant portion of the total costs, particularly for a smaller repository. When comparing the costs of RW and CO_2 disposal based simply on some sort of unit cost (i.e. costs per waste, or costs per electricity generated), this advantage might be difficult to capture. This is because, in principle, a few sites could receive all the globally generated RW, making it possible to fully realize economies of scale, whereas for the CO_2, a larger number of sites need to be explored, as the capacity of each disposal site for CO_2 is likely to be small in some geological formations compared with the amount of CO_2 generated from fossil fuel-based power plants.

3.2.3 Costs per Unit of Electricity Generation

The cost estimates from the reports reviewed above are converted into standardized units and are summarized in Table 6 to allow comparison. It has to be kept in mind that these cost estimates differ significantly in scope and coverage. Inclusion or exclusion of R&D, contingency and tax are sources of major differences. No attempt has been made here to harmonize the coverage of these cost estimates. This difficulty should be considered when comparing the numbers presented in Table 6.

All cost figures reviewed are given in overnight costs (i.e. without accounting for interest during construction and without cost escalation). Although it would be preferable to use net present values for such a comparison (e.g. discounted costs accounting for the time value of money), the published information is not detailed enough to allow the net present value to be calculated.

The cost data are first adjusted to a price level of 2000 and expressed in US dollars by applying market exchange rates. The costs are expressed in capacity units in terms of tonnes of heavy metal equivalent. Where the capacity is expressed in terms of tonnes of uranium in the original publication, conversion has been applied using the national average burnup rate and the 0.0105 coefficient explained above.

The cost data are also presented in relation to the unit of electricity generation corresponding to the amount of waste to be disposed of. The reference electricity production is not available from published studies, apart from the Japanese and Swedish studies. For all other studies, the reference electricity production corresponding to the SNF generated is estimated using the identity relating SNF generation and electricity production, as discussed in the previous section. The reference electricity production is estimated to correspond to the capacity of a repository; therefore the estimated reference electricity production may overestimate the actual electricity production, given that wastes of non-civilian origin may be included in the capacity estimates.

The amount of SNF generated per 1 GW/year of electricity production was estimated using gross thermal efficiency and the burnup ratio of each plant, and an average for a given country was calculated by weighting them with the lifetime generation as of December 2008. All data needed to estimate the amount of SNF are taken from the IAEA's PRIS database (IAEA 2010a).

For the above mentioned seven cost studies, unit costs for geological disposal of RWs range between US$113,000 and US$683,000/tHM when SNF is reprocessed and the waste comes mainly in the form of vitrified HLW, and between US$281,000 and US$650,000 when direct disposal of SNF is chosen. The costs of reprocessing and disposal of additional ILW/LLW are not included in the cost estimates for the reprocessing option. Costs for the interim storage of SNF are not taken into consideration in either case.

The Advanced Fuel Cycle Costs Basis study commissioned by the US Department of Energy (US DOE) assessed the costs of the SNF disposal at US$528,000 per tHM (in the range of US$381,000–900,000/tHM), and the costs of vitrified HLW disposal at US$211,000 (in the range of US$152,000–360,000/tHM) (at 2006 prices) (Shropshire et al. 2007).

In the SAPIERR project (Štefula 2006), international cost estimates for SNF disposal were compared. The unit cost of disposal of SNF ranges from €80,000–1,200,000/tU (at an undefined price level), with the most common values in the range of €300,000–600,000/tU (€264,000–529,000/tHM). Preliminary assessment of the data indicates the existence of the economies of scale—doubling the inventory will increase the costs (excluding the contingency and R&D expenditures) only by a factor of 1.5.

The SAPIERR II Project (Chapman et al. 2008) used linear cost models to estimate the costs for joint disposal of SNF by selected EU countries. The study is based on cost models developed by SKB (Sweden), Posiva (Finland) and Nagra (Switzerland). For each cost model, the portion of fixed costs and variable costs for several cost components was delineated. The fixed cost portions were identified as €770–1,973 million (constant December 2006 prices), and the variable costs per canister (which roughly corresponds to 2 tU) were about €650,000–880,000 (roughly €286,000–388,000/tHM).

Unit costs of RW disposal per kWh electricity generated are also computed and presented in Table 6. The unit costs are estimated to be in the range of 0.092–0.298 US cent per kWh in the case of direct disposal of SNF and of 0.036–0.221 US cent

Table 6 Summary of the unit costs of geological radioactive waste disposal

Study	Estimated total costs from Table 4	(A) Amount of SNF generated per 1 GW/year (tHM)[a]	(B) Capacity assumptions used for cost estimation (in parenthesis; estimated based on A and C) (tHM)	Estimated costs per tHM in constant 2000 US dollar prices (1,000 US$/tHM)	(C) Reference energy production (in parenthesis; estimated based on A and B) (TWh)	Estimated cost per kWh of electricity generation (in constant 2000 US cent prices)
USA	US$96.2 B (constant 2007 price)	40.2	122,100 – 111,800 (SNF) – 10,300 (HLW)[b]	650	(26,636)	0.298
Finland	€542 M (constant December 2003 price)	27.3	5,417 (SNF)[c]	427	(1,738)	0.133
Sweden	SEK 36,707 M (constant January 2003 price)	36.6	9,095 (SNF)	433	2,468	0.160
Japan	¥2,757 B (constant 2008 price)	28.3	(37,700 tHM)	683	11,670	0.221
Belgium *Reprocessing option*	€578 M (constant 2000 price)	27.5	4,697 – 4,627[c] (HLW) – 70 (MOX fuel)	113	(1,500)	0.036
Direct disposal option	€1,494 M (constant 2000 price)		4,697 – 4,097[c] (SNF) – 600[c] (HLW) – 70 (MOX fuel)	293		0.092

UK	£5,226–6,765 M (constant June 2004 price)	128.8	57,824[c] – 53,234 (HLW) – 4,590 (SNF)	133–172	(3,936)	0.195–0.252
SAPIERR (EC)	€9,029–10,662 M (constant December 2006 price)	39.5	28,800 – 25,600 (SNF) – approx. 3,200 (HLW)	281–332	(6,386)	0.127–0.150

HLW high-level waste, *SNF* spent nuclear fuel, *tHM* tonnes of heavy metal

[a] Authors' calculation

[b] The amount of commercial spent nuclear fuel stored by the end of 2007 was 57,700 tHM and the corresponding cumulative electricity production is 4,045 GW year

[c] In the respective original reports, this value was reported in tonnes of uranium. The conversion rate was calculated based on the average burnup rate for the respective country using the PRIS database

per kWh in the case of disposal of vitrified HLW. Note that as this calculation is based on non-discounted costs and non-discounted electricity production volume, the numbers are not comparable to the levelized electricity generation costs that reflect discounted costs and electricity production volumes.

The IAEA (1994) estimated the levelized unit costs of RW management and disposal for the reprocessing option (with the disposal of vitrified HLW) and for the once-through option (with disposal of SNF). Disposal costs include costs for storage and transportation, and are estimated at 0.121cent per kWh (31% of the total fuel cycle costs) and 0.192 US cent per kWh (51% of total fuel cycle costs), respectively. This was calculated using a conservative discount rate of 5% until the end of power plant life, with a zero discount rate thereafter. These costs may be compared with the cost of nuclear power electricity, which was given as 3–5 US cent per kWh at the time of publication of the 1994 IAEA report.

Finally, it is worth noting that the IEA and the OECD Nuclear Energy Agency (NEA) regularly publish levelized electricity cost estimates. In their report published in 2005 (IEA and NEA 2005) the nuclear fuel cycle cost estimates are presented for 13 OECD countries. The report distinguishes front-end and back-end fuel cycle costs, but the costs specific to the deep disposal of HLW and SNF are not provided. The back-end nuclear fuel cycle costs are in the range of US$0.07 (France) and 0.588 US cent per kWh (Japan) with the 5% discount rate, and between 0.05 and 0.479 US cent per kWh with the 10% discount rate. The prices are expressed at the 1 July 2003 level. When compared with the levelized costs of nuclear power electricity generation of the respective countries, the shares of the back-end fuel cycle costs are in the range of 2.6–12.3% with the 5% discount rate, and 1.3–7.5% with the 10% discount rate.

3.3 Calculation of Financial Liability

The IAEA Member States that signed the Joint Convention on the Safety of Spent Fuel Management and on the Safety of Radioactive Waste Management adopted basic financing principles aimed at avoiding burdens for future generations and ensuring that adequate funds are available for the proper discharge of all financial obligations for nuclear waste management (IAEA 2006).

According to the ‘polluter pays principle’, the responsibility for financing waste management lies primarily with the waste generator. In some countries, legislation mandates that the waste generator should post financial guarantees in the form of funds segregated from its normal operations. The legislative frameworks concerning financing RW management in selected OECD countries are reported to the NEA Radioactive Waste Management Committee (NEA 2003). The latest review of the status of the Joint Convention on the Safety of Spent Fuel Management and on the Safety of Radioactive Waste Management was conducted in May 2009, and it concluded that much would need to be done to meet the challenges of ensuring the availability of sufficient financial resources for effective and sustainable waste management (IAEA 2009b). Country reports are available on the IAEA's Joint Convention website (IAEA 2010b).

Many of the cost studies reviewed above were conducted to serve as the basis for deciding the contributions to waste management funds. Such cost calculations and their periodic updates are also prescribed by national legislation in some countries.

In the USA, 0.1 US cent per kWh fee is charged for civilian waste generators and deposited in the Nuclear Waste Fund. This fee does not include fees for disposal of waste from electricity generated and sold prior to 1983. The Government will share the defence part of the costs, which is, under the latest cost estimate assumptions, 19.6%. Fees collected through September 2007 totalled approximately US$21.9 billion (2007 US$) and through 2046 are expected to add a further US$19 billion (2007 US$) (OCRWM 2008c).

In Japan, NPP operators are required by law to pay contributions to the Nuclear Waste Management Organisation to cover the costs associated with the final disposal of vitrified HLW canisters. The total functional obligation was calculated as net present value of the total undiscounted cost. With a discount rate of 1.5%, the net present value of total cost was calculated as ¥1,446 billion (2008 prices), of which 39% was already paid by 2008. This total cost does not include expenditure incurred between 2000 and 2008. The contributions are paid per vitrified HLW canister, and the unit contribution is calculated and updated each year. In 2008 the contribution was set at ¥39.4 million per canister (note that the number of canisters is also discounted). This is the equivalent of ¥0.135 per kWh (approximately US$0.13 per kWh). In computing this contribution, all canisters generated in the past and in the future are taken into consideration, and the number of canisters is discounted. The fee computed in this way is more or less comparable to levelized electricity generation costs.

4 Comparing Disposal Costs of CO_2 and Radioactive Waste

This section first compares the results obtained in the previous two sections on the magnitudes and relative importance of the waste disposal costs in fossil and nuclear-based power generation. This is followed by a discussion of the main similarities and differences at the conceptual level.

4.1 Cost Comparison

The comparison is made in terms of one main indicator, the cost of disposal per unit of electricity produced. In the case of RW, this is computed by combining the cost of the minimum disposal capacity and the volume of waste to be disposed of per unit of electricity generated. There is no minimum capacity requirement for CO_2 disposal sites.

Several factors have to be taken into account when computing this indicator:

- Volume of waste to be disposed of by category: the various RW categories require different levels of safety measures that carry widely differing price tags, while CO_2 is a homogeneous waste product in this respect.
- Fuel cycle and waste management strategies: there is a choice between once-through and reprocessing cycles for RW, while no such choice exists for CO_2; fuel cycle strategies affect the volumes of HLW and SNF, whereas the choice of waste management strategy affects the volume of ILW significantly (90% volume reduction may be possible).
- Capacity of the disposal facility: for RW, the fixed cost component dominates the total disposal costs, and capacity expansion can be done at a relatively low cost; therefore the initially planned or licensed capacity of the repository is a good starting point for computing a unit investment cost for the construction of a repository. The fixed cost component is a much smaller fraction of the total disposal cost of CO_2 at any single site and, as many disposal sites are needed, the cost of CO_2 disposal is roughly proportional to the volume of CO_2 to be disposed of.
- Cost components of waste disposal are practically the same for both CO_2 and RW; they include costs related to handling the waste generated (such as pre-treatment, treatment, conditioning and transportation) and the life cycle costs of the waste disposal facility (including site exploration, engineering, operation, closure and post-closure expenditures).

A specific limitation for a meaningful cost comparison is that it is rather difficult to separate the costs strictly for disposal (establishing, operating and closing the disposal site) from the rest of the downstream fuel cycle costs of nuclear power. Yet the distortion is minor because the overwhelming share of the latter is in direct disposal costs. In contrast, by far the most expensive part of the CCD chain is capture, and transport costs are also significant.

With these caveats in mind, a comparison of the relevant columns in Tables 3 and 6 (disposal costs per unit of electricity generated for CO_2 and RW, respectively) reveals that the cost range is much smaller for RW, despite the diverging national circumstances (geological conditions, accounting rules, regulatory regimes, etc.). Skipping the reprocessing option, the costs span a range from US$0.92/MWh in Belgium to US$2.98 in the USA. There is a lot of variability in the CO_2 disposal costs, as shown in Table 3. Depending on the actual split of the US$15/t CO_2 combined transport and disposal cost in the Dooley et al. study (Dooley et al. 2008), the share of the disposal cost can vary between about 5–41% of the fuel plus disposal costs and between 1 and 14% of the LCOE.

The supply curve developed by Pöyry Energy Consulting (2007) for the UK covers a span from –€15 to €1 for the first 50 Mt CO_2 (EOR), then increases a little from €1 to €5 for the next 900 Mt CO_2 (saline aquifers), whereafter it jumps significantly to over €20 (depleted oil- and gasfields). The Eccles et al. study (Eccles et al. 2009) provides similar results for the USA.

Transport cost curves follow similar patterns. Obviously as a CCD system expands, it will first utilize low-cost combinations of transport and disposal. Over the medium and long term, the costs of other energy supply options, the prevailing CO_2 prices and other factors will determine to what extent more expensive disposal options will be used.

Because of the lack of comparable data for the different countries, it is not meaningful to attempt a quantitative comparison of the share of disposal costs in the total fuel cycle costs or in the LCOE. Yet the results in the previous two sections indicate that these shares are much larger for CCD than for RW disposal.

4.2 Selected Issues in Cost Comparison

The fundamental issue in waste disposal costs is that, to date, CO_2 emitters have been using a global public resource (the CO_2 abatement capacity of the biosphere and the global atmosphere) but will now need to shift to a similar arrangement for RW disposal that involves costs as well as using local, private or government-owned space, with some level of risk for the public. In economic terms, it was clear ever since the beginning of nuclear energy programmes that the costs of the safe management and final disposal of RW must be part of internal or private costs. This is because the health and environmental impacts of RW were never considered as a candidate for externality, and the possibility of compensation for damages was only raised in the case of unintended/accidental release of radioactive material from the RW management process. CO_2 has been vented from the burning of fossil fuels for centuries; its negative environmental impacts through the modification of the climate system have been understood only in the past few decades. This issue, together with the need to reduce emissions and compensate for climate change damage, has been raised only relatively recently in various international forums. This means that fossil fuel burners will need to internalize these external costs by paying for CCD themselves, purchasing CO_2 emission permits, paying the applicable carbon tax or not to operate at all. The first three cases imply a significant new cost element in fossil-based electricity costs, while disposal costs have always been included in nuclear power in one form or another.

An important difference is the related regulatory frameworks and the resulting decisions based on cost implications. Strict regulations for handling and disposing of RW have been in place for decades to minimize any inadvertent external effects from the release of radioactive material. New regulation will be required for internalizing the climate externality of CO_2. Investment decisions about CCD will depend on the nature of the regulation and the resulting carbon price. A command-and-control type technology standard (no new coal-fired power plants to be permitted without CCD) would bring some degree of certainty in terms of emissions, but the related costs might be high. A carbon tax would provide an input for deciding whether to build new fossil plants with CCD or just capture-ready (hedging against future carbon tax increases), while an emission permit trading scheme and the inherently large uncertainties about future carbon prices would make fossil power investment decisions even more difficult. Technology standards are the only regulatory option for RW. The associated costs may be high but there are no external sources of uncertainty influencing the decision as to whether or not they should be borne, whereas the market-based regulation of carbon prices heavily influences the decision about adding CCD to fossil plants.

A related issue with significant cost implications is whether leakage from the disposal site can be tolerated or not. Van der Zwaan and Gerlagh (2008) analyse the economic aspects of CCD in relation to the possibility of significant leakage of CO_2 from geological reservoirs. They review the economic and climatic implications of the large-scale use of CCD for reaching a stringent climate change control target when geological CO_2 leakage is accounted for. Their model includes three main CO_2 mitigation options: energy savings, transition to non-carbon energy sources and the use of CCD. The authors find CCD to be a valuable option, even with CO_2 leakage of a few per cent per year, well above the maximum seepage rates foreseen by geological assessments. However, this analysis focuses on the atmospheric and climate implications of CO_2 leakage and does not account for the potential environmental impacts, human health and economic damages at the local/regional scale where the leakage occurs. The possibility of leakage here is rather different from the case of RW, in which no leakage of radioactive material is tolerated over very long time horizons—practically until the level of radioactivity declines to that of natural uranium—except in rare cases in which sufficient dilution can be proven. The cost difference between imperfect and nearly perfect containment can be significant.

Both CO_2 and RW disposal involve a series of legal-economic issues that are linked to the ownership of the underground disposal space. Depending on the legal system, subterranean space may belong to the owner of the surface land area or to the community (government). For RW it is possible in principle to secure, at a relatively low cost, the ownership of the total surface area under which the disposal facility is constructed and operated because of the limited surface area required for even a large depository. As CO_2 disperses over large distances in the disposal media from the injection wells, this would be rather difficult for CO_2 technically and extremely expensive economically. A hitherto totally ignored aspect in CCD cost estimates is the price of using someone else's underground property in the first place and, more importantly, compensating the owner for making it unusable for any other purpose for a very long time (option value). This could be a particularly contentious issue in the case of disposal sites spanning national borders unless a joint operation is agreed between the states in question.

Irrespective of property rights, a related economic issue is the notion of underground space as a depletable resource and the associated scarcity rent. The ratio between the amount of SNF and HLW arising from even an extremely large-scale nuclear power expansion with a once-through fuel cycle (an unlikely scenario in itself) and the volume of geologically suitable space for their disposal is so low that the question of disposal space scarcity is irrelevant. In contrast, even optimistically assessed potential CO_2 disposal space would not be able to accept more than a few decades' (perhaps a century's) worth of CO_2 produced (although disposal space and thus the fill-up rate is highly dependent on the region) and as suitable disposal space becomes depleted, so the remaining space would have an increasing scarcity value. Yet currently this issue seems to be of conceptual interest only, as payment for using underground space is virtually absent from the existing literature.

Using up a finite resource raises the question of possible backstop technologies. This involves yet another difference, at least in the narrow sense. Irrespective of whether scarcity rents will or will not be reflected in disposal costs, with the depletion of suitable disposal space the only backstop technology for CO_2 is mineralization, which is a very energy-intensive and thus rather expensive process. If disposal space were ever raised as a limiting factor for RW, closing the fuel cycle with fast reactors burning minor actinides would be a technically feasible solution that, among other benefits, would reduce the volume and radiotoxicity of the ultimate waste products. In a broader sense, other power generation technologies or system solutions (e.g. smart grids with myriads of decentralized electricity storage capacities) might emerge as a backstop for both fossil and nuclear electricity if they can provide the same level and reliability of service at a lower price.

With a view to financing disposal costs, the most important difference in the economics of RW and CO_2 disposal concerns timing. In the case of RW it is possible to accumulate the funds necessary for all disposal-related costs as part of the operating costs from electricity sales because, in a given fuel cycle arrangement, the waste volume is proportional to the electricity generated. Moreover, it is safe and inexpensive to store SNF and HLW for decades until the disposal facility is established (acknowledging the undeniable ethical concerns and the existence of some risks). During this time the accumulating disposal fund can even earn interest. No fund accumulation option exists for CO_2. All capture facilities, transport lines and disposal sites must be put in place before the first molecules of CO_2 can be prevented from entering the atmosphere. This involves a need to finance all related investments and the corresponding costs of capital (interest during construction, etc.) before the CO_2 benefits can be harvested. For new coal power plants this means a significant increase in investment costs compared to the non-CCD alternative, increasing their share in total LCOE and thereby approaching the cost structure of nuclear electricity. Yet CCD also has a significant operating cost component related to the energy required for the capture and conditioning of CO_2.

The potential for using waste as a resource involves similarities and differences. Although the main objective of the geological disposal of both CO_2 and RW is to isolate these substances from the rest of the biosphere, at least a part of them could be used as a productive resource. CO_2 can be an indirect resource in EOR, EGR and in ECBM recovery for mobilizing economically valuable resources (oil, natural gas, methane) while RW itself is a potential resource as long as it contains material that can be separated and used in nuclear reactors. However, only a small portion of the capturable CO_2 can be used productively because of geological and economic considerations (how much CO_2 is needed and what the cost-benefit ratio is of transporting CO_2 to distant EOR or EGR sites), whereas most of the RW remains a potential resource for a long time. This leads to a major difference in the requirements for retrievability: it is sometimes a regulatory requirement to make the retrieval of RW possible for at least 100 years for possible reuse (or for improved disposal if better packaging material for encapsulation or advanced disposal technologies become available), while retrieval is at best an option for remediation in the case of CO_2 leakage from the disposal site (see Maul 2011).

The prevalence of uncertainty in the disposal cost assessments is a common feature for CO_2 and RW. Lacking any industry-scale full-chain CCD demonstration facility, cost estimates to date are derived from similar industrial processes (e.g. the oil and gas sector) and from experiments with separate components of the CCD chain, and remain rather speculative. The longer history of R&D, including the construction of underground research laboratories, provides some basis for estimating RW disposal costs. However, a considerable degree of uncertainty remains about the costs of all necessary materials, equipment, surface and underground facilities required for the full-scale operation of repositories.

Another common feature is that, in the midst of the prevailing uncertainty of cost estimates, cost will vary widely across countries and regions, mainly because of site-specific conditions and partly driven by the efficiency of the regulatory and implementing organizations.

Both fossil fuel and nuclear electricity generation involve a certain public good characteristic to the extent they enhance energy supply security. Having them in the electricity generation portfolio increases the diversity of supply. The use of coal from domestic or reliable foreign sources as well as availability of uranium from many politically stable regions and the competitive fuel industry make both technologies a secure source. The economic value of this public good is not reflected in the price of electricity but should be considered when public resources need to be made available for disposing of the related waste (e.g. providing underground space or direct financial support).

Another similarity is related to economic competitiveness. Fossil-based electricity and nuclear power compete with each other and with other electricity technologies. The costs of CCD and RW disposal are important factors in the competitive position of both technologies. However, the relationship between waste disposal costs and competitiveness is often blurred by various government interventions (special taxes or subsidies, explicit or hidden) with even greater impacts on competitiveness.

The stability of the regulatory system is crucial for both technologies. They involve expensive long-lived capital assets. Once these are in place, it would be a major economic loss not to use them at full capacity, let alone to retire them prematurely.

In summary, there are some similarities concerning the costs of CO_2 and RW disposal that can provide a basis for preparing in-depth analyses on specific issues, like the value of stable regulation and the extent to which it can foster relevant investments. Nonetheless, major differences dominate this comparative assessment: these range from the need to pay for waste disposal (an obvious element in the cost of nuclear power as opposed to a newly emerging cost item for fossil-based electricity) to the physical scarcity of the disposal space and the issue of accounting for scarcity rent in the cost calculations.

This section has also revealed the difficulties of framing meaningful comparisons of the disposal costs for CO_2 and RW. Despite all the caveats about accounting differences between CCD and RW disposal, and also within each domain, some might find it interesting to compare the disposal costs per unit of electricity generated. However, what really counts in public and private decision making is the LCOE that includes all investment, financing, fuel, operation and maintenance, waste disposal and decommissioning costs in the nation-specific geographic, natural resource, economic and political context.

5 Summary and Conclusions

In the final account: direct geological disposal costs are a small fraction of the LCOE for nuclear power and also for fossil fuel electricity. Accounting for the preceding steps (capture, conditioning and transport) and considering the total downstream fuel cycle costs, however, the full CCD component triggers a major increase in the fossil-based electricity cost, while remaining very small for nuclear power.

Insights from comparing the disposal costs per se are limited for the above reasons. The total downstream costs differ more significantly both in absolute terms (per MWh) and relative terms (as a share of the total electricity cost), with CCD being a much larger cost item in both instances. What ultimately counts in economic terms is LCOE together with all external costs (remaining CO_2 emissions, radiation risks, etc.). Yet the simple comparison exercise presented in this chapter might help find the most critical elements in the fuel cycles in which technological improvements could lead to cost reductions and thus enhance the competitiveness of the respective technologies.

The most profound difference in the costing of CO_2 and RW disposal is that the former represents a completely new cost item in fossil fuel electricity, whereas the latter has been an obvious item on the cost sheet since the 1960s, irrespective of whether the corresponding fee was collected and accumulated during the operation of NPPs or not. Accounting for CO_2 disposal costs and especially for the other related downstream costs (capture and transport) will trigger a significant increase in LCOE generated from the burning of coal or gas.

The other fundamental difference with severe implications for the disposal costs and thus for the LCOE stems from the timing of the investment into waste disposal relative to the time of the power generation. RW can be stored safely at low cost for decades before emplacement into the final repository, and this leaves ample time to accumulate the disposal costs by charging a small fee per unit of electricity generated. In contrast, CO_2 abatement requires immediate disposal after capture because temporary storage would be prohibitively expensive. Therefore, the investment portion of the disposal (as well as capture and transport) costs must be disbursed before CCD operation can commence.

The two waste management technologies share important regulatory concerns with cost implications. Sloppy or frequently changing rules, standards and other regulatory elements trigger significant increases in the disposal costs and increase the cost of capital because of uncertainties, as well as discouraging private investors. Therefore, clear and concise policies translated into stable and reliable regulation are crucial for both technologies.

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