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Two-scale evaluation of remediation technologies for a contaminated site by applying economic input–output life cycle assessment: Risk–cost, risk–energy consumption and risk–CO_2 emission

Yasushi Inoue*, Arata Katayama

EcoTopia Science Institute, Nagoya University, F3-4(670) Furo-Cho, Chikusa, Nagoya 4648603, Japan

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ABSTRACT

A two-scale evaluation concept of remediation technologies for a contaminated site was expanded by introducing life cycle costing (LCC) and economic input–output life cycle assessment (EIO-LCA). The expanded evaluation index, the rescue number for soil (RN_{SOIL}) with LCC and EIO-LCA, comprises two scales, such as risk–cost, risk–energy consumption or risk–CO_2 emission of a remediation. The effectiveness of RN_{SOIL} with LCC and EIO-LCA was examined in a typical contamination and remediation scenario in which dieldrin contaminated an agricultural field. Remediation was simulated using four technologies: disposal, high temperature thermal desorption, biopile and landfarming. Energy consumption and CO_2 emission were determined from a life cycle inventory analysis using monetary-based intensity based on an input–output table. The values of RN_{SOIL} based on risk–cost, risk–energy consumption and risk–CO_2 emission were calculated, and then rankings of the candidates were compiled according to RN_{SOIL} values. A comparison between three rankings showed the different ranking orders. The existence of differences in ranking order indicates that the scales would not have reciprocal compatibility for two-scale evaluation and that each scale should be used independently. The RN_{SOIL} with LCA will be helpful in selecting a technology, provided an appropriate scale is determined.

1. Introduction

To facilitate decision making in selecting a remediation technology for a contaminated site, a comprehensive multiscale evaluation method is required to rank the applicable alternatives according to various factors with a trade-off relationship, such as risk reduction, cost, environmental impact, resource consumption and other factors. We have proposed a two-scale, risk–cost evaluation method for remediation technologies – the so-called rescue number for soil (RN_{SOIL}) [1]. The two scales are the residual contaminant risk and total remediation cost. The essence of this conceptual method is to reveal the increase in economic cost, environmental impact or resource depletion accompanied with the reduction of contaminant risk (i.e., a difference between initial risk and residual risk) when a remediation is implemented, and to examine a balance with risk reduction. RN_{SOIL} is calculated to determine the ranking of remediation technologies using estimated indicators. Although the two-scale evaluation was developed conceptually, the appropriate estimation method of the indicator was not sufficiently advanced to acquire a correct total cost; rather, it enabled only a rough estimation based on the average unit cost of remediation technologies, and it might lead to under- or overestimation for newly developed technologies. The indicator needs to be estimated prospectively and to indicate site-specific characteristics.

In order to help stakeholders, especially residents or landowners, to make a decision regarding the selection of an appropriate option, rapid and simple methodology for estimating the indicators is required. Life cycle costing (LCC) is one of the methods that can be used to estimate total cost prospectively based on all the remediation processes [2]. LCC will provide the total cost estimated with higher accuracy than that based on mean unit cost, reflecting site-specific characteristics of remediation. In addition to the total cost, the energy consumption could be a useful indicator for the trade-off with risk reduction. Furthermore, the environmental impact is also an important factor to analyze in the trade-off with risk reduction. The amount of carbon dioxide (CO_2), one of the major environmental burdens, could be proportional to energy consumption because it is considered a good indicator of energy consumption from human activity.

* Corresponding author. Tel.: +81 52 7895858; fax: +81 52 7895857.
E-mail addresses: inoue@esi.nagoya-u.ac.jp (Y. Inoue), a-katayama@esi.nagoya-u.ac.jp (A. Katayama).

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Many researchers claim that life cycle assessment (LCA) plays an important role in estimating the emissions and the environmental impact of various human activities. LCA has increasingly been used to support site remediation decision making by estimating environmental impact and resource consumption [3,4]. Researchers have compared contaminant risk with environmental impact associated with remedial activities using risk assessment and prospective LCA [5–8]. Other researchers have estimated potential environmental and health impacts associated with remediation by conducting LCA in order to show that effects of remediation extend to local, regional and global scales [9,10]. A few researchers have conducted only life cycle inventory (LCI) analysis to estimate some inventories of emissions on remediation, and then interpreted them to compare remedial activities using standardized scores from emissions fluxes [11] or impact units from concentration limits [12]. According to these studies, prospective LCA would be useful to estimate environmental impacts associated with the remediation of soil contamination, but much information would be required for an impact assessment. When life cycle costs have been obtained, economic input–output life cycle assessment (EIO-LCA) is available for the rapid estimation of inventories of emissions [13]. Some inventories are readily estimated from the price of materials or services and the emission intensities determined by the input–output analysis. A combination of LCC and EIO-LCA would be useful for estimating three indicators – total cost, energy consumption and CO_2 emission – as used in RN_{SOIL}.

The objective of this study is to expand the concept of the two-scale evaluation, RN_{SOIL}, by introducing LCC and EIO-LCA, to determine indicators and to examine the feasibility of using RN_{SOIL} with EIO-LCA for establishing technology for the screening of level ranking. A virtual scenario of contamination and applicable remediation was used as an example, and LCC and LCI analyses were carried out to determine total costs, followed by a determination of energy consumption and CO_2 emission, as one of the two scales. Three definitions of RN_{SOIL} with EIO-LCA were determined, and technologies were ranked according to their performance.

2. Methods

2.1. Two-scale evaluation method, RN_{SOIL}

The RN_{SOIL} is composed of two originally defined indices: figure of treatment priority (*FTP*) representing priority of treatment, and figure of unprocessibility for waste (*FUW*) representing difficulty of treatment [14]. *FTP*, a numerical risk index, has been defined as the number of people exposed to a contaminant in amounts exceeding the acceptable daily intake (ADI) of the chemical concerned, considering the daily human health risk caused by chronic toxicity of the contaminant in a specific area. *FUW*, a numerical cost index, has been defined as the total cost of completing the remediation technology. In this study, *FUW* was redefined as any one indicator among energy consumption or CO_2 emission, in addition to total cost. RN_{SOIL} has been defined as the product of the time integration of *FTP* and *FUW*, without any weighting, as expressed by Eq. (1). It represents the immaturity and competitiveness of a remediation technology through a comparative evaluation:

$$RN_{SOIL} = \int_0^{t_L} FTP(t)dt \cdot FUW \quad (1)$$

where t_L is the period of exposure (e.g., lifetime) and $FTP(t)$ means the time-varying *FTP*. Time integration of $FTP(t)$ indicates cumulative residual risk and it can demonstrate the difference in risk reduction and duration of remediation between technologies. Smaller values of RN_{SOIL} indicate a better technology. A complementary index indicating the effectiveness of remediation, rescue index (*RI*), has been defined to support the selection. Reduced risk, represented by the difference between time integration of $FTP(t)$ for implementation of remediation ($FTP_R(t)$) and time integration of $FTP(t)$ for no remedial action ($FTP_O(t)$), is used in the definition of *RI* as follows:

$$RI = \frac{\int_0^{t_L} \{FTP_O(t) - FTP_R(t)\}dt}{FUW} \quad (2)$$

where the subscripts *O* and *R* represent the implementation of remediation and no remediation, respectively. *RI* indicates the efficiency of technology – a larger *RI* value indicates higher efficiency [1].

2.2. Scenarios of soil contamination and remediation

For the hypothetical scenario used in this simulation, we assumed farmland contaminated with dieldrin – one of the persistent organic pollutants. Dieldrin was formerly widely used as a pesticide, although since 1970 its production and use have been prohibited in Japan because of its high toxicity and persistency in the environment [15]. The pollutant was buried in the ground near agricultural fields and forests, and left there until an efficient treatment technology was established. Recently, in Japan, dieldrin has been detected in agricultural field soil and cucumbers grown there [16]. The requirement to treat buried dieldrin and soil contaminated with dieldrin under the Stockholm Convention on Persistent Organic Pollutants took effect in 2004 [15]. In our scenario, the farmland is located near an urban area. The pollutant contaminated an area of 1000 m^2 (approximately 31.6-m^2) to a depth of 0.5 m. Uniform contaminant concentration was assumed to be 1 mg/kg soil. The surrounding area and contaminated site had quite a large population density (4179.6 people/km^2) [17]. The period considered in the risk assessment was 70 years from the start of remediation. The ADI was 0.0001 mg/kg/day [18].

Four technologies – disposal, high temperature thermal desorption (HTTD), biopile and landfarming – were selected to achieve the soil guideline level (0.001 mg/kg soil). Table 1 lists the processes and properties of the technologies used for conducting the LCI analysis.

Disposal and HTTD are off-site technologies that require transport of the removed soil by dump trucks. In our scenario, the quantity of soil excavated was 1000 m^3: 500 m^3 of contaminated soil and 500 m^3 of lower layer soil. The distance from the contaminated site to the disposal facility and the thermal treatment plant was assumed to be 10 km. Residual contaminated soil emitted by the HTTD was dumped at the disposal facility. The distance from the HTTD plant to the disposal facility was also assumed to be 10 km.

Biopile is an on-site technology, for which soil piles are required to maintain the degrading ability by aeration and tillage. Landfarming is also an on-site technology, for which excavation is not required. For both biopile and landfarming, the half-life of dieldrin in the soil was assumed to be 180 days, as estimated from the degradation rate of dieldrin-degradable white-rot fungus [19]. This half-life was used to determine the remediation period.

2.3. Quantification of FTP

Temporal changes of *FTP* for all scenarios were determined using the method of Inoue and Katayama [1] as follows. $FTP(t)$ was defined as Eq. (3):

$$FTP(t) = \int_{ADI}^{+\infty} f(E(t))dE \times pA \quad (3)$$

Table 1
Properties and processes of the test technologies.

Technology	Disposal	High temperature thermal desorption (HTTD)	Biopile	Landfarming
Place of treatment	Off-site	Off-site	On-site (with excavation)	On-site (without excavation)
Treatment rate	220 m^3 soil/day[a]	3 metric tonnes soil/h[b]	Half-life: 180 days[c]	Half-life: 180 days[c]
Processes	Temporary enclosure Excavation Dust reduction Drainage treatment Monitoring Backfilling and recovery of soil function Transport of soil Disposal	Temporary enclosure Excavation Dust reduction Drainage treatment Monitoring Backfilling and recovery of soil function Transport of soil Thermal desorption (off-site)	Temporary enclosure Excavation Dust reduction Drainage treatment Backfilling and recovery of soil function Construction of piles Maintenance Monitoring	Incubation Transport of culture Spreading culture solution Biodegradation Agitation of soil Monitoring
Period of remediation	15 days	32 days	60 months	60 months
Unit cost from published databases[d]	48,000–81,300 yen/m^3[e]	8,100–25,200 yen/m^3	13,000–26,000 yen/m^3	<10,000 yen/m^3

a [19].
b [20].
c [21].
d [22]; costs in yen were calculated from original data in US dollars using 100 as the dollar–yen conversion rate.
e Data were converted to values per cubic meter from original values per metric tonnes, using 1.6 metric tonnes/m^3 as bulk density.

where f shows a probability density function of human exposure, E, p is population density and A is the local area. Only chronic toxicity and noncarcinogenic effect on humans were considered; acute toxicity to humans and ecological toxicity were not included. In this hypothetical case study, the main exposure route to human beings was ingestion of agricultural products. The probability density distributions of daily human exposure were calculated using the multimedia fate, transport and exposure model CalTOX (DTSC, California EPA, USA), combined with the Crystal Ball (Kozo Keikaku Engineering, Tokyo, Japan), for the Monte Carlo simulation, assuming mean and standard deviation values of the model parameters with log-normal distribution. Almost all the parameters in the CalTOX model used default values. Area of site, depth of soil layer (surface, root zone, vadose zone and aquifer), annual average precipitation, land area fraction of surface water area, and reaction half-life in soil and groundwater were set corresponding to the scenarios. Average body weight was taken as 50 kg. The probabilities of daily exposure exceeding the ADI were calculated from the probability density distribution of exposure. Temporal changes in $FTP(t)$ were then obtained as the number of individuals by multiplying the probability by the number of people who lived in the area adjacent to the target area. $FTP(t)$ was simply an index to show differences in the contamination risk between remediation technologies clearly. All scenarios were examined under the following assumptions: the site continued to be used as an agricultural field and people ingested the agricultural products from the site.

2.4. Quantification of FUW

The functional unit of LCC and LCI analysis is "to reduce a contaminant concentration below the soil criterion for 1000 m^3 soil of agricultural field". In some literature, the system used for LCC and LCI analysis was designed from soil surveillance as an anterior temporal boundary to completion of remediation as a posterior temporal boundary [9,10]. In the present study, because the aim of the LCC and LCI was to compare the applicable technologies for a specific site, a system boundary was determined from installation of a temporary enclosure to the recovery of soil fertility. Hence, the on-site process, the off-site process and the regeneration of soil fertility after completion of remediation were included in the system. Surveillance of contaminant distribution was excluded from the system because this process was common to all alternatives. Raw materials and energy acquisition, monitoring and waste management were included in each process. The entire process was divided into several subprocesses, and the materials, equipment, fuel and labor required in each process were quantified in detail. Quantities and unit prices were obtained from technical documents [20–22]. Based on items in the entire process, LCC was conducted to estimate cumulative cost by integrating the costs for all items and subprocesses. The cumulative cost from LCC was defined as total cost (C). The unit prices of materials, fuel, water and labor were determined from the actual market prices [23]. Properties of the thermal desorption process in the HTTD, such as treatment rate, quantities of materials and specifications of equipment, were obtained from technical data [24]. For biopile, the cost-estimating program of NAVFAC ESC [25] was used. For nondisposable and repeatedly used equipment and machinery, including trucks, backhoes and pumps, the cost of usage was estimated by a depreciation cost over the remediation period, using a list of depreciation cost calculations for construction machinery [26]. The fuel consumption rate of heavy machinery and the number of workers were determined based on the public work estimation standards [20,22]. Labor cost was estimated using a standard unit price for labor [27].

For prospective LCI analysis, we used the spreadsheet "Embodied Energy and Emission Intensity Data for Japan using Input–Output Tables" (3EID), which is very useful for estimating energy consumption and CO_2 emission based on the producer price of materials or processes [28]. CO_2 emission as an indicator included only CO_2, and energy consumption included both fossil energy and renewable energy. In brief, if we have an inventory of cost, then an inventory of energy consumption and of CO_2 emission can be readily estimated using this worksheet. Embodied environmental intensity in 1995, taking import products into consideration, was applied. The most likely sector in the input–output table was selected for each item used for descriptive purposes. For example, as shown in Table 2, "metal products for construction" were selected from the categories in 3EID for temporary-use materials for site enclosure. The representative energy consumption and CO_2 emission intensity used are listed in Table 2. The CO_2 emission intensity for diesel oil and electricity was obtained from Japan Society of Civil Engineers [29]. For machinery, multiplying the emission intensity by the depreciation price for the in-service period gave the CO_2 emission [30]. Energy consumption and CO_2 emission from labor were estimated by applying the method proposed by Kusuda

Table 2
Energy consumption intensity and CO_2 emission intensity (partial).

Item	Energy consumption intensity (TOE/million yen)	CO_2 emission intensity (metric tonnes CO_2/million yen)	Categories in 3EID[a]
Temporary use materials, iron sheets and pipes	1.843[a]	6.550[a]	Metal products for construction
Backhoe and bulldozer	1.188[a]	3.933[a]	Mining, civil engineering and construction machinery
Dump truck	0.9698[a]	2.843[a]	Trucks, buses and other cars
Sheets and pipes	1.802[a]	4.747[a]	Plastic products
Turbidity water treatment	1.238[a]	4.134[a]	Pumps and compressors
Generator	1.149[a]	3.753[a]	Engines
Analysis	0.7028[a]	1.762[a]	Research and development (intra-enterprise)
Soil disposal	0.6654[a]	7.542[a]	Waste disposal services (industrial)
Labor	0.001073[b]	0.01338[c]	–
Diesel oil	0.9126[a]	0.0779[d]	Diesel oil
Electricity	86[a]	0.129[d]	Electricity

TOE, metric tonnes oil equivalent (1 TOE = 10^7 kcal).
Labor had a unit in terms of TOE/person/day, diesel oil: TOE/kL, electricity: TOE/10^6 kWh.
Labor had a unit in metric tonnes CO_2/person/day, diesel oil: kg C/L, electricity: kg C/kWh.
[a] [29]; 3EID represents "Embodied Energy and Emission Intensity Data for Japan using Input–Output Tables".
[b] Calculated from the energy consumption in the residential sector in Japan [33].
[c] Estimated using the method of Kusuda [31].
[d] [32].

[31]. Based on the yearly CO_2 emissions from a domestic source [32], the Japanese population [33] and daily working time (8 h), the hourly CO_2 emission per person was calculated. Finally, LCI analysis yielded the total cost (C), energy consumption and CO_2 emission for four remediation scenarios. In order to examine the accuracy of total cost (C), total cost (U) was defined as total cost calculated from the actual performance unit cost and calculated using maximum values in the range of unit costs from a published database, as shown in Table 1. The unit costs based on American prices were assumed to be applicable to the same remediation technologies implemented under Japanese conditions. The differences in unit costs between technologies were also assumed to be the same as under Japanese conditions. The economic loss due to inoperative periods of the site was not included in the remediation cost.

2.5. Calculation of RN_{SOIL} and RI, and illustration of a risk–cost diagram

The evaluation indices RN_{SOIL} and RI were calculated from the estimated $FTP(t)$ and FUW values using Eqs. (1) and (2). All $FTP(t)$ values of the remediation technologies were integrated over a period of 70 years, independent of the remediation period. In the case of no remedial action, $FTP_O(t)$ was calculated and integrated to obtain RI. Corresponding to the different definitions of FUW as total cost, energy consumption and CO_2 emission, three different sets of evaluation indices were calculated. Diagrams for risk–cost, risk–energy consumption and risk–CO_2 emission, with one axis for the cumulative residual risk index (integral $FTP(t)$) and the other axis for the cost index (FUW), were used to characterize the remediation technologies. With the aim of supporting the selection of the most appropriate remediation technologies, the relative superiority between pairs of technologies was determined, based on the risk–cost diagram [1]. Four remediation scenarios were plotted and their orders of rank for the three different FUWs were determined.

3. Results and discussion

3.1. FUW

The type of contaminant makes little difference for physicochemical treatments, but biological technologies are heavily dependent on the bioavailability of the contaminant. The rate of decrease of dieldrin in the biodegradation would be much lower than that of hydrocarbons, because the bioavailability of dieldrin is fairly low [34]. Therefore, for dieldrin-targeted remediation technologies, the actual performance data of mean price leads to an underestimation of total cost, and the total cost (C) by LCI analysis is correctly evaluated by considering the length of the remediation period. Fig. 1 shows the total cost (C) of four remediation technologies. Disposal had the highest total cost (C), followed by biopile, landfarming and HTTD. Under the scenario in this study, LCI analysis can provide a better estimation of total cost as total cost (C) through LCC, because the total cost (C) of biopile and landfarming was significantly larger than the total cost (U) calculated from the mean unit cost of technology.

Figs. 2 and 3 show the energy consumption and the CO_2 emission of the remediation scenarios, respectively. In terms of energy consumption, HTTD had the largest consumption, followed by disposal, biopile and landfarming. In addition, in terms of CO_2 emission, disposal had the largest emission, followed by HTTD, biopile and landfarming. The difference between the biological and physicochemical technologies was greater for CO_2 emission than for energy consumption. Although energy consumption and CO_2 emission were estimated based on total cost results, they were not proportional to the total cost. The reason why HTTD had lower CO_2

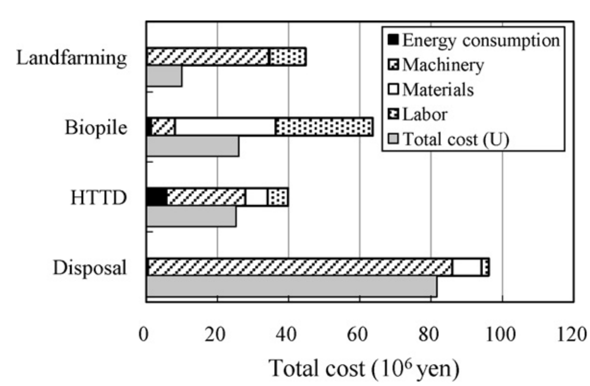

Fig. 1. Total cost (C) of technologies calculated from inventory data. Total cost (U) of technologies from unit cost based on actual performance data is also indicated.

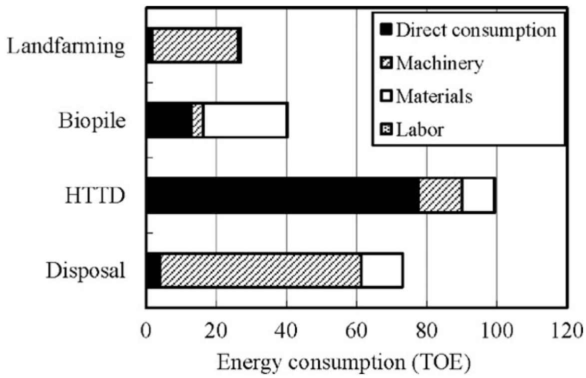

Fig. 2. Energy consumption of technologies calculated from inventory data.

emission than disposal, in spite of the highest energy consumption, is the high CO_2 intensity and high cost due to extra disposal fees. The CO_2 intensity was significantly larger (7.542 metric tonnes CO_2/million yen) than that of the other items, as listed in Table 2. In order to prevent the risk of pollution, most disposal facilities in Japan tend to avoid the dumping of contaminated soils into their disposal facilities. Extra fees are required for the disposal of contaminated soil, thus the disposal costs of contaminated soil are usually very high in Japan.

Energy consumption per cost unit was 0.604, 0.635, 2.50 and 0.760 TOE/10^6 yen and CO_2 emissions per cost unit were 1.57, 1.80, 8.71 and 7.13 metric tonnes CO_2/10^6 yen for landfarming, biopile, HTTD and disposal, respectively. The data indicate that biological technologies had an advantage over physicochemical technologies in terms of CO_2 emission. There was about a twofold difference in the total cost between disposal (highest) and HTTD (lowest). On the other hand, there was about a ninefold difference in the CO_2 emission between disposal (highest) and landfarming (lowest), and about a 2.7-fold difference in energy consumption between HTTD (highest) and landfarming (lowest). This indicated that these three scales do not have reciprocal compatibility, and should be used as independent factors for a technological ranking.

For the purpose of highlighting the technological characteristics, *FUW*s were divided into four areas: energy consumption, machinery used, materials consumed and labor. Energy consumption includes all energy used during the remediation process. Machinery only includes the production of materials; it does not include direct energy use in the process. HTTD had higher direct energy consumption than the other technologies on all scales because of high fuel (heavy fuel oil) consumption in the thermal desorption process. Direct energy consumption of HTTD comprised 78% of the total consumption, and was more than the total energy consumption of the other technologies. Machinery used in disposal was >80% on all scales because of its high cost (value was determined from actual maximum price; 35,000 yen/m^3 soil) and quite a large emission intensity (7.542 tonnes CO_2/million yen) in the soil disposal process, as shown in Table 2. The transportation process occupied less than 10% in the total of every *FUW*. The long distances to thermal plants and landfill sites resulted in increased *FUW*s (total cost, energy consumption and CO_2 emission). A 10-times greater distance than a standard scenario (10 km) would increase every *FUW* by at most 10%. Disposal resulted in at least twice as much CO_2 emission as from HTTD, and seven times as much as for biological technologies. Biopile had the largest amount of materials consumed on all scales because of the requirement for piles constructed on-site. Machinery used in landfarming occupied a great portion of the total on all scales because of scheduled soil agitation for long periods. The labor costs of biopile and landfarming were higher than those of the physicochemical technologies because of the longer remediation periods and the requirement for much more manpower for scheduled maintenance and monitoring of chemicals. However, labor contributed little to energy consumption and CO_2 emission in all scenarios. This resulted in the large difference between the biological and physicochemical technologies in terms of energy consumption and CO_2 emission. In these scenarios, technological characteristics were reflected appropriately in the inventories.

When considering improvements to the technologies, LCI analysis will be useful to show which process will be the most effective for reducing the *FUW*. Landfarming and biopile had a high ratio of labor in the total cost, but these processes would be ineffective for reducing energy consumption and CO_2 emission. A longer remediation period leads to an increase in not only labor cost but also in machinery used (in landfarming) and in materials consumed (in biopile). The machinery in landfarming was mainly used in soil agitation, and the materials in biopile were consumed for maintenance and monitoring. Reducing these indicators would require a reduction in the remediation period, that is, an enhancement of the degradation rate. By contrast, HTTD had a high ratio of direct energy consumption and disposal had a high ratio of machinery. Direct energy consumption in HTTD was dominated by the operation of thermal desorption equipment, that is, fuel consumption in a heating furnace. Greater effectiveness in thermal desorption is required to reduce the fuel consumption. CO_2 emission from machinery was dominated by the machinery used in the soil disposal process. The reduction of a large amount of soil to be disposed is required for achieving an improvement in disposal.

3.2. Two-scale comprehensive evaluation index, RN_{SOIL}

For the four remediation technologies, Figs. 4–6 show the relationship between the cumulative residual risk and the total cost (Fig. 4), the energy consumption (Fig. 5) and the CO_2 emission (Fig. 6) in the scenario applied in this study. Disposal, HTTD and biopile, which all involve a soil excavation process, had a relatively low level of cumulative residual risk, and there were no great differences between them in this respect. This is because the excavated soil was transported or piled, and contaminants were managed to prevent volatilization into the air or infiltration into the ground. By contrast, the cumulative residual risk of landfarming was >40 times that of the other remediation technologies. This is because landfarming involves no soil excavation or removal and contaminants persisted in the site.

In the figures, constant RN_{SOIL} can be drawn as a straight line with a minus one slope. Disposal was assumed to be the benchmark technology, and the line indicating the constant RN_{SOIL} of disposal

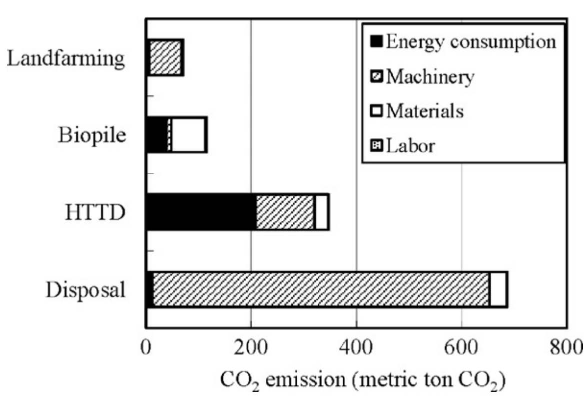

Fig. 3. . CO_2 emission of technologies calculated from inventory data.

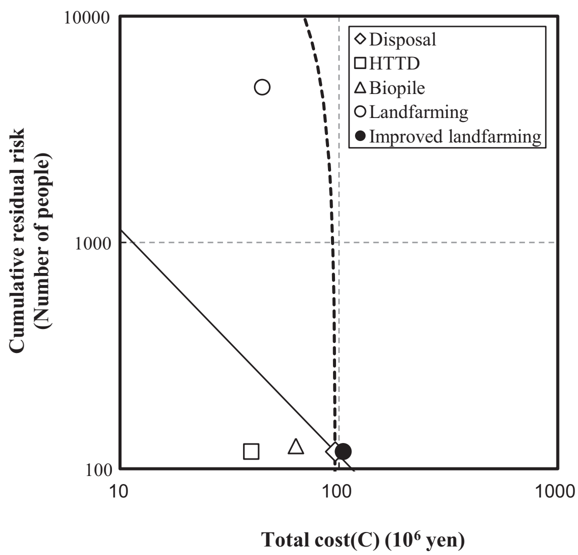

Fig. 4. Relationship between residual risk and total cost (C) of technologies. The solid line and dotted line show the constant values of RN_{SOIL} and RI for disposal as the benchmark, respectively. Improved landfarming is also plotted as a solid circle.

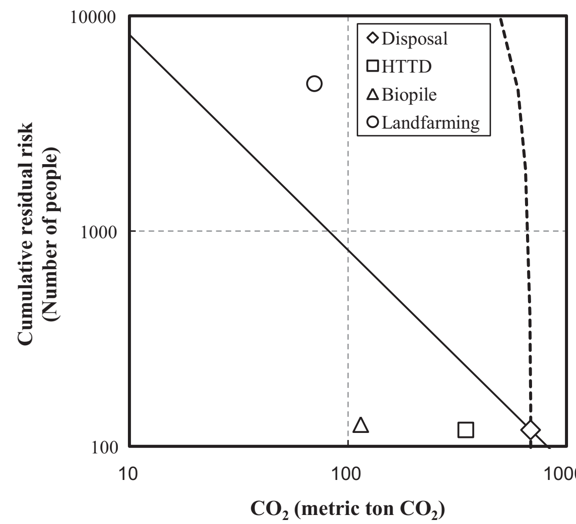

Fig. 6. Relationship between residual risk and CO_2 emission of technologies. The solid line and dotted line show the constant values of RN_{SOIL} and RI for disposal as the benchmark, respectively.

was drawn to aid the comparison with the other technologies. The technology plotted below this straight line is evaluated as being higher than the benchmark, whereas the technology plotted above this line is evaluated as being lower than the benchmark. From these three figures, it was found that the rankings differed from each other. Based on total cost, the highest ranking technology was HTTD, followed by biopile, disposal and landfarming (Fig. 4). Biopile had the highest ranking in the RN_{SOIL} based on energy consumption, followed by disposal, HTTD and then landfarming (Fig. 5). Biopile had also the highest ranking in the RN_{SOIL} based on CO_2 emission, followed by HTTD, disposal and then landfarming (Fig. 6). Landfarming had a great cumulative residual risk, which led to it being the lowest ranked. This suggested that the superiority of landfarming in terms of energy consumption and CO_2 emission (Figs 2 and 3) did not compensate for its inferiority in terms of cumulative residual risk.

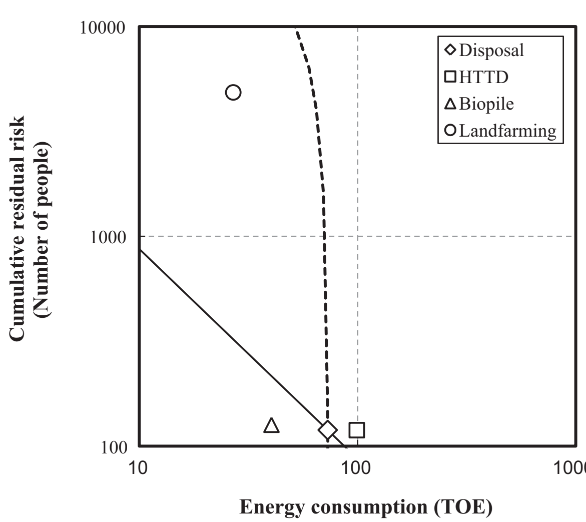

Fig. 5. Relationship between residual risk and energy consumption of technologies. The solid line and dotted line show the constant values of RN_{SOIL} and RI for disposal as the benchmark, respectively.

Table 3 shows the values of RN_{SOIL} and RI based on total cost (C), energy consumption and CO_2 emission. The ratios of test technologies in the RN_{SOIL} and RI to the benchmark as a technological difference are also shown in this table. In the case of RN_{SOIL}, the degree of difference between technologies varied greatly from scale to scale. Total cost (C) had the largest difference and CO_2 emission had the smallest difference between benchmark and landfarming (worst). The order of ranking in the RI was different from that of the RN_{SOIL}. There were trends that biological technologies had higher ranking in the RI, for example, landfarming had the highest (energy consumption and CO_2 emission) or second highest ranking (total cost). Two evaluating indices, RN_{SOIL} and RI, will make the technological ranking more definite.

These results indicate that the performance of landfarming needs to be improved if it is to compete with the other technologies. Values of indices would be useful as targets for technological improvement, and a method for achieving target values in landfarming will be discussed below. Here we examine landfarming in terms of total cost (C) as an example of the interpretation of the results. To achieve competitiveness with the benchmark technology (disposal) based on total cost (C), the RN_{SOIL} for landfarming has to be reduced to 1/19 of its current value to reach the same value as the benchmark (Table 3). An effective way to reduce the RN_{SOIL} is to apply an extra process, decreasing the cumulative residual risk of landfarming. One of the extra processes, namely, tight migration control of dieldrin, can be introduced by insulating the site from the surrounding environment with a temporary structure and using treatment equipment for an exhaust gas and water containing dieldrin. Tight migration control was estimated to be able to reduce the cumulative residual risk by up to 1/40 of the current value. On the contrary, total cost (C) was estimated to increase to 105 million yen by adding at least 60 million yen for the construction of a temporary structure and the operation of treatment equipment. Improved landfarming only appeared to approach the benchmark; it would be difficult to exceed it (as plotted in Fig. 4). Only once the migration control of dieldrin was successfully introduced as an additional process, at a cost of 95 million yen (within increase of 50 million yen above original total cost), would landfarming become competitive to the benchmark.

For the other scales, such as energy consumption and CO_2 emission, a similar examination can be carried out to improve test

Table 3
Comparison of RNSOIL values for total cost, energy consumption and CO_2 emission.

Indices	Scales for *FUW*	Disposal (benchmark)	HTTD	Biopile	Landfarming
RN_{SOIL}	Total cost (C) ($\times10^{10}$)[a]	1.15 (1.0)	0.474 (0.41)	0.803 (0.70)	21.7 (19)
	Energy consumption ($\times10^4$)[b]	0.871 (1.0)	1.18 (1.4)	0.509 (0.58)	13.1 (15)
	CO_2 emission ($\times10^4$)[c]	8.17 (1.0)	4.13 (0.51)	1.44 (0.18)	34.1 (4.2)
RI	Total cost (C) ($\times10^{-4}$)[d]	3.65 (1.0)	8.83 (2.4)	5.52 (1.5)	6.79 (1.9)
	Energy consumption ($\times10^2$)[e]	4.80 (1.0)	3.53 (0.74)	8.69 (1.8)	11.2 (2.3)
	CO_2 emission ($\times10^2$)[f]	0.512 (1.0)	1.01 (2.0)	3.07 (6.0)	4.33 (8.5)

Values in parentheses refer to the ratio to the benchmark (disposal).
[a] Unit for RN_{SOIL} (cost) is number of people × yen.
[b] Unit for RN_{SOIL} (energy) is number of people × TOE.
[c] Unit for RN_{SOIL} (CO_2) is number of people × metric tonnes.
[d] Unit for *RI* (cost) is number of people/yen.
[e] Unit for *RI* (energy) is number of people/TOE.
[f] Unit for *RI* (CO_2) is number of people/metric tonnes.

technologies. Competitiveness must be evaluated from a multiscale consideration of the trade-off relationship. To consider a decrease in the RN_{SOIL} to the same value as the benchmark, we need to examine whether a degree of decrease in the cumulative residual risk falls below a degree of increase in the total cost, energy consumption or CO_2 emission.

3.3. Usefulness of the RN_{SOIL} based on EIO-LCA

Technological ranking using the RN_{SOIL} values was determined by means of risk assessment and EIO-LCA, based on the concept of a trade-off relationship between risk reduction and increases of the indicators (total cost, energy consumption and CO_2 emission). EIO-LCA is an inexpensive method, which uses publicly available data and enables rapid analysis. It is appropriate for screening level analysis [35]. Therefore, RN_{SOIL} based on EIO-LCA offers a great advantage in that less effort, time and money are required. In the decision making at the first screening level, with limited information on remediation technologies, the RN_{SOIL} based on EIO-LCA will provide a ranking of remediation technologies. Because residents or local government usually do not have sufficient and adequately detailed information on remediation technologies, EIO-LCA would be the preferred methodology for estimating indicators, rather than process-based LCA.

Although EIO-LCA offers several advantages, as discussed above, a significant extent of uncertainty remains regarding the RN_{SOIL} value. Mainly two types of uncertainty in data existed: uncertainty in intensities from the EIO-LCA model and uncertainty in technological characteristics. The uncertainty in intensities affects indicators (energy consumption and CO_2 emission for each process) in EIO-LCA. Uncertainty in the EIO-LCA model is derived from old data, uncertainty inherent in the original data, incomplete original data, aggregated original data, aggregation of sectors and other factors [12]. Thus, uncertainty of emission intensities causes the indicators (energy consumption and CO_2) and the RN_{SOIL} values to be significantly uncertain.

The uncertainty in technological characteristics affects the determination of total cost via LCC. Some properties expressing remediation technology, such as treatment rate and transportation distance, affect a remediation period, resulting in a change in total cost. The excavation rate in excavation disposal and HTTD was determined by a Japanese governmental organization for estimating public work costs; hence, there should be a relatively small uncertainty here. The treatment rate of thermal desorption on HTTD would be affected by the size of rotary kiln as the thermal furnace and the water content in the soil. The uncertainty would be smallest compared with other technologies; hence, the process was fully implemented under controlled conditions. Because the quantity of soil to be treated was not great, the uncertainty of the treatment rate would little affect the necessary periods of excavation disposal and HTTD. In contrast, no performance data for biotechnologies, biopile and landfarming for dieldrin contamination were available. These technologies were completely hypothetical scenarios. Treatment rates of biodegradation were determined from one experimental result in the literature [21]. Literature relating to the dieldrin degradation rate reveals that it has a wide distribution (or half-life of dieldrin) under various conditions, including laboratory and field experiments (at least one order of spread in the half-life). Treatment rates of biodegradation had relatively large uncertainties. Biopile was implemented under controlled conditions on-site, so the uncertainty would be smaller compared with landfarming. Much longer remediation periods of biopile and landfarming would make uncertainties in the indicators and RN_{SOIL} larger. The periods of remediation were within the range 15 days to 60 months. The economic loss due to these inoperative periods of the site would affect the selection of technology. On-site biological technology will be strongly affected, and lead to a lower evaluation in terms of technological ranking. Eventually, uncertainties of treatment rate would affect the order of the RN_{SOIL} values, especially in landfarming.

Because EIO-LCA leads to unavoidable uncertainty, as discussed above, EIO-LCA is unsuitable for a close investigation. Developers or consultants who have detailed technological information can however conduct process-based LCA to obtain indicator values (emission inventories). Process-based LCA will be suitable to compare their own technology with that of other companies and to improve it by process analysis. However, because process-based LCA is data intensive and requires comprehensive analysis [36,37], determination of the RN_{SOIL} with process-based LCA is time consuming and expensive to conduct. Selection of LCA methodology should be based on who uses the index and/or for what purpose.

The most appropriate indicator used in RN_{SOIL} should be selected to correspond to a difference in values or interests of each stakeholder. The RN_{SOIL} based on total cost will be valuable for the person who is liable for the cost of remediation or for the general public as taxpayers (in the case of the use of public funds). The RN_{SOIL} based on CO_2 emission or energy consumption will be useful for the person or group who has strong environmental awareness, especially groups that are harmed by specific impacts due to the emission. The difference in the order of ranking according to the indicators, as found in this study, shows that the selection of an indicator to be compared with risk reduction is important. A detailed discussion of the difference in LCA methodology should be investigated based on comparison of the results of EIO-LCA with results of process-based LCA.

4. Conclusions

The concept of RN_{SOIL} for remediation technology was successfully expanded by introducing LCC and EIO-LCA. The concept of RN_{SOIL} with EIO-LCA shows the relative competitiveness of several alternative remediation technologies, and it can be used to rank them roughly, but rapidly, for the purpose of decision making by nonexperts (residents, land owners or local government) in the screening stage. Multiple-scale estimation based on LCC and LCI analysis led to a clear indication of the differences between technologies in terms of different characteristics. The ranking of the technologies differed on each scale. The overall suggested rankings are as follows. Technology without soil excavation and transport (landfarming) is inferior to the other technologies in the RN_{SOIL} because of the greater exposure. Biological technology (biopile) is superior in the RN_{SOIL} based on both the energy consumption scale and the CO_2 emission scale. Degradation or separation technologies (biopile or HTTD) are superior in the RN_{SOIL} based on the cost scale. Technology with high direct energy consumption (HTTD) is inferior in the RN_{SOIL} based on the energy consumption scale.

Factors contributing to different rankings were the higher intensity of direct energy consumption and lower intensity of labor, in comparison to the other inventory items, on both the energy consumption and the CO_2 emission scales. The greater CO_2 emission intensity of the soil disposal process also contributed to the difference in ranking. Because this ranking was for rough screening, some extent of uncertainty in estimated indicators (energy consumption and CO_2 emission) was acceptable.

The existence of differences in the order of ranking for each scale indicates that these scales will not have reciprocal compatibility for two-scale evaluation. A priority scale will first have to be decided upon and an appropriate scale selected as the *FUW* indicator. A technological improvement was examined, based on the technological rankings and the differences of indicators between technologies. Because the proposed method is used only for the screening level of nonexperts in the initial stage of risk communication, if the RN_{SOIL} is established by conducting a process-based LCA and following environmental impact assessment, it will contribute to optimization, by highlighting an effective point that needs improvement, as a developer support tool. Further case studies on an actual remedial site are required to verify the evaluation.

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References

[1] Y. Inoue, A. Katayama, Application of the rescue number to the evaluation of remediation technologies for contaminated ground, J. Mater. Cycles Waste Manage. 6 (2004) 48–57.
[2] Y.S. Sherif, W.J. Kolarik, Life cycle costing: concept and practice, Omega 9 (3) (1981) 287–296.
[3] G. Lemming, M.Z. Hauschild, P.L. Bjerg, Life cycle assessment of soil and groundwater remediation technologies: literature review, Int. J. Life Cycle Asses. 15 (1) (2010) 115–127.
[4] S.A. Morais, C.A. Delerue-Matos, Perspective on LCA application in site remediation services: critical review of challenges, J. Hazard. Mater. 175 (1–3) (2010) 12–22.
[5] A. Blanc, H. Métivier-Pignon, R. Gourdon, P. Rousseaux, Life cycle assessment as a tool for controlling the development of technical activities: application to the remediation of a site contaminated by sulfur, Adv. Environ. Res. 8 (2004) 613–627.
[6] S. Volkwein, H.W. Hurtig, W. Klöpffer, Life cycle assessment of contaminated sites remediation, Int. J. Life Cycle Asses. 4 (5) (1999) 263–274.
[7] G. Lemming, P. Friis-Hansen, P.L. Bjerg, Risk-based economic decision analysis of remediation options at a PCE-contaminated site, J. Environ. Manage. 91 (5) (2010) 1169–1182.
[8] G. Lemming, M.Z. Hauschild, J. Chambon, P.J. Binning, C. Bulle, M. Margni, P.L. Bjerg, Environmental impacts of remediation of a trichloroethene-contaminated site: life cycle assessment of remediation alternatives, Environ. Sci. Technol. 44 (23) (2010) 9163–9169.
[9] M.L. Diamond, C.A. Page, M. Campbell, S. McKenna, R. Lall, Life-cycle framework for assessment of site remediation options: method and generic survey, Environ. Toxicol. Chem. 18 (4) (1999) 788–800.
[10] C.A. Page, M.L. Diamond, M. Campbell, S. McKenna, Life-cycle framework for assessment of site remediation options: case study, Environ. Toxicol. Chem. 18 (4) (1999) 801–810.
[11] L. Toffoletto, L. Deschênes, R. Samson, LCA of ex-situ bioremediation of diesel-contaminated soil, Int. J. Life Cycle Asses. 10 (6) (2005) 406–416.
[12] R.P. Vignes, Use limited life-cycle analysis for environmental decision-making, Chem. Eng. Prog. 97 (2) (2001) 40–54.
[13] C.T. Hendrickson, L.B. Lave, H.S. Matthews, Environmental Life Cycle Assessment of Goods and Services: An Input–output Approach, Resources for the Future Press, Washington, DC, 2006.
[14] C. Yamauchi, H. Ito, T. Fujisawa, H. Matsuda, A. Katayama, T. Tsunekawa, T. Saito, T. Kitsuka, T. Tanaka, K. Ishihara, Evaluation system for advanced waste and emission management, Waste Manage. Res. 12 (3) (2001) 183–186 (in Japanese).
[15] Ministry of the Environment in Japan (MOE), The National Implementation Plan of Japan Under the Stockholm Convention on Persistent Organic Pollutants. Available from: http://www.env.go.jp/chemi/pops/plan/all.pdf, 2005 (accessed 20.06.10) (in Japanese).
[16] Y. Hashimoto, Dieldrin residue in the soil and cucumber from agricultural field in Tokyo, J. Pestic. Sci. 30 (4) (2005) 397–402.
[17] Ministry of Agriculture, Forestry and Fisheries in Japan (MAFF), World Agricultural Census 2000, vol. 9, Report on Rural Communities. Available from: http://www.e-stat.go.jp/SG1/estat/List.do?bid=000001013531&cycode=0, 2000 (accessed 03.12.03) (in Japanese).
[18] World Health Organization (WHO), Inventory of IPCS and Other WHO Pesticide Evaluations and Summary of Toxicological Evaluations Performed by the Joint Meeting on Pesticide Residues (JMPR) Through 2009. Available from: http://www.who.int/ipcs/publications/jmpr/jmpr_pesticide/en/index.html, 2009 (accessed 10.11.15).
[19] Ministry of Land, Infrastructure, Transport and Tourism in Japan (MLIT), Standard Labor Cost of Public Work, 2004 ed. Available from: http://www.mlit.go.jp/kisha/kisha05/01/010329_2_.html, 2005 (accessed 17.04.05) (in Japanese).
[20] Naval Facilities Engineering Service Center (NAVFAC ESC), Application Guide for Thermal Desorption Systems, Technical Report TR-2090-ENV, Port Hueneme, California, 1998.
[21] D.W. Kennedy, S.D. Aust, J.A. Bumpus, Comparative biodegradation of alkyl halide insecticides by the white rod fungus, *Phanerochaete chrysosporium* (BKM-F-1767), Appl. Environ. Microbiol. 56 (8) (1990) 2347–2353.
[22] Federal Remediation Technologies Roundtable (FRTR), Remediation Technologies Screening Matrix and Reference Guide Version 4.0. Available from: http://www.frtr.gov/matrix2/top_page.html, 2007 (accessed 16.11.07).
[23] Ministry of Land, Infrastructure, Transport and Tourism in Japan (MLIT), Civil Engineering Work Estimation Standard, 2004 ed., Construction Research Institute, Tokyo, 2004 (in Japanese).
[24] Construction Research Institute (CRI), Manuals for Estimation Standard of Civil Engineering Works, 2004 ed., Construction Research Institute, Tokyo, 2004 (in Japanese).
[25] Construction Research Institute (CRI), Standard Price of Estimation on Civil Engineering Works, 2004 ed., Construction Research Institute, Tokyo, 2004 (in Japanese).
[26] Construction Research Institute (CRI), Prices of Construction Materials and Wages, Construction Research Institute, Tokyo, 2005 (in Japanese).
[27] Naval Facilities Engineering Service Center (NAVFAC ESC), Biopile Cost Estimating Program Version 3. Available from: http://enviro.nfesc.navy.mil/erb/erb_a/restoration/technologies/remed/bio/BPCE%20.xls, 2000 (accessed 02.11.05).
[28] Japan Construction Mechanization Association (JCMA), List of Depreciation of Machineries for Construction, 2005 ed., Japan Construction Mechanization Association, Tokyo, 2005 (in Japanese).
[29] National Institute for Environmental Studies in Japan (NIES), Embodied Energy and Emission Intensity Data for Japan Using Input–Output Tables, Center for Global Environmental Research. Available from: http://www.cger.nies.go.jp/publications/report/d031/eng/table/download/1995/ei/ei95399p_e.xls, 2002 (accessed 09.08.09).
[30] Japanese Society of Civil Engineering (JSCE), Committee of Global Environment: Report on Research in 1995, 1995 (in Japanese).
[31] T. Kusuda, LCA on public water supply system, in: H. Imura (Ed.), LCA on Construction, Ohmsha, Tokyo, 2001, pp. 177–186 (in Japanese).
[32] Ministry of Internal Affairs and Communications in Japan (MIC), Statistical Data/Demographic Shifts. Statistics Bureau. Available from: http://www.stat.go.jp/data/jinsui/2004np/index.htm, 2004 (accessed 27.01.05) (in Japanese).

[33] Ministry of the Environment in Japan (MOE), Global Environment Bureau: Preliminary Figures of Global Warming Gas Emission in 2004, 2005 (in Japanese).
[34] A. Katayama, Pesticide degradation by soil microorganisms, in: Japanese Society of Soil Science and Plant Nutrition (Ed.), Environmental Restoration by Plants and Microorganisms, Hakuyusya, Tokyo, 2000, pp. 125–153 (in Japanese).
[35] M. Bilec, R. Ries, H.S. Matthews, A.L. Sharrard, Examples of a hybrid life cycle assessment of construction processes, J. Infrastruct. Syst. 12 (2006) 207–215.
[36] M.A. Curran, Environmental Life-cycle Assessment, McGraw-Hill, New York, 1996.
[37] J.R. Deanna, A.B. Fullerton, Industrial Ecology: US–Japan Perspectives, National Academy, Washington, DC, 1994.