

Chapter 2

Life-cycle Cost Approach (LCCA): Framework and Concepts

2.1 Framework

As discussed, life-cycle cost approach (LCCA) has evolved from a project appraisal tool to a more comprehensive method of incorporating sustainable development aspects in various sectors. LCCA could be conceived in the broader sustainable development framework. The framework consists of three interconnected sustainability dimensions, such as economic, environmental and social. Economic sustainability concept draws from the public finance framework using financial and economic assessment of investments. Environmental sustainability is based on externalities framework (again from 'public good' and public finance). Social sustainability draws from public policy framework where service delivery, governance and social equity are critical. Achieving sustainability on these three counts is a challenge. The nexus approach of water, energy and food security (Hoff 2011) comes close to addressing this challenge. The nexus approach provides a broader framework within which granularity exists. Here, granularity is referred to in the linkages within the sector and sub-sectors, for instance, within water sector, the linkages between surface and groundwater resources, between irrigation and drinking water. Similarly, within drinking water, the linkages between water, sanitation, wastewater, reuse of wastewater, etc., are very much interlinked organically. The granularity is well captured in the three overarching questions raised by Kurian and Ardakanian (2013), (i) intersectionality (critical mass of factors at the intersection of material fluxes); (ii) interactionality (interactions with exogenous factors, viz., policy, economy, environment, etc.; and (iii) hybridity (building transdisciplinary approaches).

Life-cycle thinking is the conceptual idea behind LCCA that reflects the comprehensiveness of the approach in a systems perspective. LCCA takes the whole chain and spread of activities that take into consideration the nexus and

the embedded granularity. It takes all the phases of the life cycle of a product or service that are required during pre-production, production and post-production into consideration. These include even the externalities of the production process (Fig. 2.1). It is also argued that the applicability of LCCA in development projects is limited in scope in the context of developing countries, as the all-pervasive social and political drivers are not adequately considered in the present LCCA tools (McConville 2006). LCCA is also data intensive, often making it difficult to use for development work. A life-cycle evaluation of development projects must incorporate diverse factors in a practical manner with a judicious mix of quantitative and qualitative aspects. Further, lack of formal guidelines and reliable past data and difficulty in estimating future costs appear to be the main reasons for the tardy adoption of LCCA. The tool, therefore, must be consistent with successful development practices and simplified for use as a common tool. This could be achieved through a combination of methods and tools for understanding the dynamics.

Though LCCA has potential to deal with various externalities associated with the process, it is not possible to include and assess all the externalities associated with the process of production of any goods and services. While it is easy to scope (consequential) the externalities, it is not easy to assess the impact of these externalities (attributional). It is therefore necessary to define the system boundaries in order to reduce the complexity of assessing the impacts of all the externalities

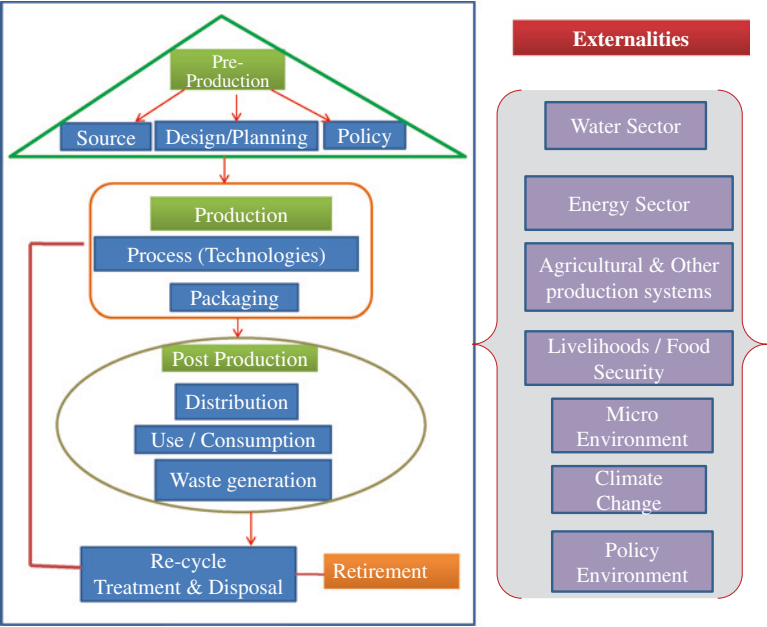


Fig. 2.1 LCCA framework in nexus approach

associated with the process. The choice of system boundaries depends on the nature and type of the product or service in question, which would have important implications on the results (Lundin 2002) and needs to be carefully considered. The life-cycle (or functional) boundaries define the processes to be included in the system, i.e. where upstream and downstream cut-offs are set. Functional boundaries limit the various aspects that are to be included for the assessment. These are mainly related to the environmental externalities. There are three major types of system boundaries: between the technical system and the environment, between significant and insignificant processes, and between the technological system under study and other technological systems (Guinée et al. 2002, as quoted in Finnveden et al. 2009).

Here, we present a generic LCCA framework that shows the possible phases of processes of product or service. These phases could be considered as system boundaries in a simplified version. At each phase, system boundaries can be a set of complex interlinkages. In this generic framework, we look at four phases and the system boundaries (Fig. 2.1). Pre-production phase (level 1) boundaries are defined to ensure resource sustainability and make judicious design and planning for sustainability. The assessment at this level helps in understanding potential environmental issues associated with basic source (raw material extraction). The designing and planning for the production phase is also included and needs to incorporate these costs in conjunction with the policies.

The second phase pertains to production where the emphasis is on infrastructure, technologies and is usually linked to the management agency/institution/organisation. This provides a more complete view of the system in terms of technologies, design efficiencies, planning (viz. linking products and by-products) and packaging. Often the agencies, though aware, are constrained by financial and legislative obligations and tend to override options that allow for a move towards environmental sustainability in the production phase. They either may adopt partially or may not adopt at all. Such a perspective may limit the potential of the agency to identify major environmental impacts or improvements through the life cycle.

The third phase deals with the post-production issues that are often dealt at the community/institutional/household level. These pertain to use/consumption (domestic, agriculture, industry, etc.), and use practices, including waste generation, reuse, recycling, treatment and disposal. This can happen at the production phase as well. And ultimately the retirement of the uneconomic infrastructure. Often, this set gets marginal attention, if not ignored, at the project planning level. This set reflects and determines the adoptability to the system in terms of capacities (technologies), affordability (finance), awareness (quality, health, etc.), attitudes (cultural), etc.

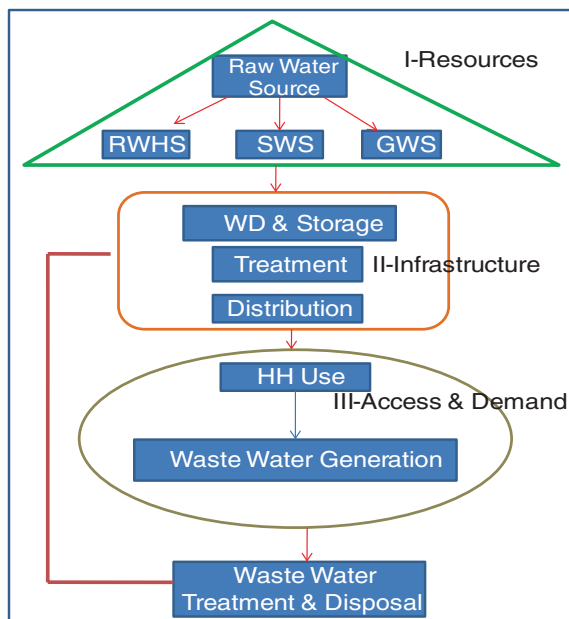
The fourth phase represents the externalities of or to the system that is closely linked and surrounding the main system. The sustainability dimension of LCCA lies in capturing and assessing these externalities. Surrounding systems interact and are critical for the functioning of the core system. Water,

energy and land are critical to any production system. While they are often factors of production and included in the costs, these systems also are affected in the production process. Such costs or benefits need to be taken into account. Agriculture production or farming systems (including forestry, livestock, etc.) determine not only demand for the products or services (fertiliser, pesticides, water, etc.) but are also affected in the process (land degradation, chemical use, etc.). These processes would affect the microenvironment in the case of waste or effluent discharge and affect livelihoods positively as well as negatively. Other important factors like climate and policy changes add the risk and uncertainty dimension to the whole process. These need to be taken into account while assessing costs.

This framework can be articulated in the context of water and sanitation that are mostly dependent on scarce groundwater resources in developing countries. Groundwater is exploited for the purpose of supplying drinking water in rural and urban areas. These resources are neither protected from overexploitation nor supported through replenishing mechanisms (like percolation tanks, etc.). There are competing demands for water from agriculture, industry and other livelihoods. In most cases, there are no policies to address these issues. This is part of the pre-production phase, where one has to include the costs of not only identifying and locating the resource but also include costs of planning and design for their sustainable use in the end. During the production phase, different technologies are used to exploit, treat and distribute the water. Here, identifying appropriate technologies that provide optimum benefits is necessary for financial sustainability of the system. Besides, managing the infrastructure is critical for maintaining the life of the infrastructure and sustaining the services. Energy sector plays a critical role at this phase. During the post-production phase, distribution and use are critical for social sustainability in terms of attaining equity in the distribution of service. Here, the institutional and governance aspects play an important role in ensuring social sustainability. Reuse, recycling, treatment and disposal are important for environmental sustainability. Wastewater generated from water, sanitation and hygiene (WASH) services in the urban areas is used for irrigating crops in the peri-urban areas. While the use of wastewater provides livelihoods and economic benefits to communities, it also results in negative impacts like water quality deterioration, health impacts, human as well as livestock, etc. (Reddy and Kurian 2010). Apart from these externalities, the linkages between groundwater and energy also result in externalities such as resource degradation. These externalities can be internalised with judicious planning. The problems of degradation further aggravate in the context of climate variability or policy distortions. Policies like free power would increase the risk of degradation.

In the context of life-cycle costing (LCC), the system boundaries are limited to economic or financial costs. The costs of infrastructure and distribution are only included (Fig. 2.2). Even the use-level costs are not included in the case of financial costs, though economic costs include user costs.

Fig. 2.2 LCC system boundaries for drinking water supplies



2.2 Cost Components

LCCA analyses the aggregate costs through the life cycle of the system or infrastructure. In a standard LCCA, acquisition costs and sustaining costs are included at the aggregate level (Barringer 2003). These costs are also termed as recurring and non-recurring costs or fixed and variable costs. Each of these costs will have various components of costs at the disaggregate level. Acquisition costs include hardware and software costs. Hardware costs include mainly infrastructure, buildings, etc., while software costs include research and design costs, capacity building, etc. Broadly, the cost components include not only the construction and operational costs but also the rehabilitation and information, education and communication (IEC) costs. These are as follows: capital expenditure on hardware (initial construction cost) (CapExHrd); capital expenditure on software (CapExSoft); capital maintenance expenditure (rehabilitation cost or CapManEx); cost of capital (CoC); direct support costs (ExDS); indirect support costs (ExIDS); and annual operation and maintenance cost (OpEx). These are broadly grouped under fixed and recurring costs (Box 1).

While fixed costs include source protection and construction (hardware) along with designing and planning (software). Variable or recurring costs include capital or asset maintenance; operation and maintenance costs, CoC, direct and indirect support costs, including training, planning and institutional support. The delivery of sustainable services also requires that financial systems be in place

in order to ensure that infrastructure can be renewed or replaced at the end of its useful life and to extend delivery systems in response to increases in demand (Reddy et al. 2009).

Depending on the nature of the product or service, it is likely that households, apart from public utilities or private agencies, also invest or incur costs. These costs could be fixed or variable depending on the product or service. It is observed that households often spend substantial amounts towards fixed and variable costs in order to improve the WASH service provided by public agencies, viz. infrastructure such as wells, storage, toilets, etc., and operational costs such as minor repairs, cleaning, etc. These costs are incurred in order to overcome reliability and convenience issues related to water services. Along with these expenditures, households also spend time fetching water and money towards buying water. These are incurred to overcome access and quality problems. While monetary expenditure alone is considered in the case of financial analysis, economic analysis includes both public and household expenditure in monetary terms, as well as opportunity costs. On the other hand, in case of sanitation, public and household expenditure are mutually inclusive, as household expenditure is a necessity and mandatory for construction of household toilets. Hence, both public and household expenditures need to be analysed together for sanitation.

Another set of costs that are important in a comprehensive life-cycle cost analysis (green economy approach) are the costs associated with environmental externalities. These include degradation costs of natural resources like soil, water, air, etc.; emissions or effluents that directly affect livelihoods, health, etc.; and long-term impacts like greenhouse gases (GHGs), etc. These impacts could be positive or negative. They could take place within the sector or product that is being assessed or any other sector linked to the core sector.

Box 1: Cost Components

Fixed Costs

CapExHrd: this includes government expenditure on infrastructure such as water sources, pumps, storage, filters, distributions systems, etc.

HHCapExHrd: this includes household expenditure on infrastructure such as water storage, toilets, wells, pumps, etc.

CapExSft: this includes government expenditure on planning and designing costs of the schemes.

Recurring Costs

CapManEx: this includes capital maintenance such as rehabilitation of sources, systems, etc.

CoC: this includes the interest paid on the borrowed capital for investment in the WASH sector.

ExDS: this includes staff salaries, post-implementation activities such as IEC, demand management and training of mechanics.

ExIDS: *this includes policy planning at the macro level, i.e. central and state.*

OpEx: *this includes regular operation and maintenance of the systems such as energy costs, minor repairs, filtering costs, salaries of water man, etc.*

HHOpEx: *this includes household expenditure on operation and maintenance of water systems, sanitation facilities, etc.*

RTCOST: *Retirement costs include the termination costs of the infrastructure.*

Costs of Environmental Externalities

These include resource degradation costs within sector and in other sectors that are linked to the core sector.

2.3 Discount Rates, Annualisation and Functional Unit

All the fixed capital investments are made over the years and are hence cumulated over the years. Similarly, benefit flows are cumulated over the years. When LCCA is adopted at the initial stages of the project, the capital or fixed investments are made in the current year and the recurring investments are made in future years over the life of the system. Some of these costs are regular and expected (operation and maintenance), and others could be irregular and unexpected (capital maintenance). Benefit flows take place in future years. In order to make project appraisals comparable between products or services, all these costs and benefits need to be assessed at the current year. In cases where LCCA is taken up at a later stage of the project, historical costs and benefits are used where costs and benefits would accrue in the past as well as in future. These costs and benefits are inflated to the current year level. Various deflators (future benefits) or inflators (past investments and benefits) are suggested in the literature (Barringer and Weber 1996). These range from the National GDP inflator/deflator (inflation based) to fixed consumption (depreciation) and accelerated depreciation or appreciation. In the case of environmental benefits, lower discount rates are often proposed (Table 2.1).

Different systems have different lifespans, including technical, economic and useful. In order to make the projects comparable, the lifespans need to be standardised by annualising the costs. In order to arrive at the unit costs per year, all the capital costs (CapExHrd) are annualised using the normative lifespans of the systems, i.e. the technical lifespan. Arriving at the lifespan of a system becomes complicated where different components of the system have different lifespans. Using component-wise lifespans for hardware such as boreholes, pumps, pump houses, overhead reservoirs, hand pumps, etc., is more realistic. While normative lifespan is determined technically, it may not hold well in reality. Systems may last longer or shorter than their normative life due to various factors such as poor maintenance and natural factors like hydrogeology; precipitation, temperature and humidity; and natural disasters like floods, droughts, etc. The actual lifespan is the actual number of years the component lasts. By comparing these two, one can

Table 2.1 Present value of investments using GDP inflator for four countries (1961–2010)

	1961	1962	1963	1964	1965	1966	1967	1968	1969	1970	1971	1972	1973	1974	1975
India	100	102	104	109	111	111	113	108	108	115	117	121	130	134	148
Burkina Faso	100	102	104	109	111	111	113	108	108	115	117	121	130	134	148
Ghana	100	103	105	113	124	145	157	151	170	188	194	204	235	284	354
Mozambique															
1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	
160	171	203	233	251	273	310	338	356	379	382	358	362	374	393	
160	171	203	233	251	273	310	338	356	379	382	358	362	374	393	
458	587	982	1,702	2,347	3,547	6,230	7,968	17,774	24,050	29,016	41,118	57,237	76,355	97,960	
					100	104	122	138	163	217	244	687	1,019	1,501	
1991	1992	1993		1994	1995	1996	1997	1998	1999	2000					
400	384	385		379	435	464	466	473	510	533					
400	384	385		379	435	464	466	473	510	533					
128,490	154,242	171,440		225,884	293,941	420,468	587,973	702,382	822,128	936,989					
2,013	3,242	4,377		6,387	9,946	15,036	24,787	27,011	28,468	29,714					
2001	2002	2003		2004	2005	2006	2007	2008	2009	2010					
524	545	579		580	603	601	601	623	659	680					
524	545	579		580	603	601	601	623	659	680					
1,192,132	1,607,207	1,973,949		2,540,560	2,905,134	3,339,850	6,036,783	7,019,369	8,437,429	9,848,758					
33,289	38,242	41,438		43,600	46,858	50,972	55,722	59,836	64,834	68,254					

Source: Provided by Mr. Peter Burr, WASHCost Project, IRC, The Netherlands.

assess whether the actual cost of provision is more or less than the estimated costs. Moreover, actual lifespan takes into account the risk and uncertainty associated with the system.

Standardisation is also necessary for comparing the environmental benefits or disbenefits. Functional units are specified for each assessment, and they should be comparable across the products or services, for instance, emissions per unit (kg) of product or wastewater generated per unit of water in filtering (litres).

2.4 Components of Life-cycle Cost Model

The basic LCCA functional form should include the components as indicated in Eq. 2.1.

$$LCC_{xt} = f \left\{ \sum_{t=1}^n (\text{CapEx}_{hw_{xt}}; \text{CapEx}_{sw_{xt}}; \text{CapManEx}_{xt}; \text{CoCap}_{xt}; \text{DsCost}_{xt}; \text{IDsCost}_{xt}; \text{OpEx}_{xt}) + \text{CoEExt}_{xt} \right\} \quad (2.1)$$

where

LCC_{xt}	Life-cycle costs of specified product/service
$\text{CapEx}_{hw_{xt}}$	Capital expenditure on hardware (initial construction cost)
$\text{CapEx}_{sw_{xt}}$	Capital expenditure on software
CapManEx_{xt}	Capital management expenditure (rehabilitation cost)
CoCap_{xt}	Cost of capital
DsCost_{xt}	Direct support costs
IDsCost_{xt}	Indirect support costs
OpEx_{xt}	Annual operation and maintenance cost
CoEExt_{xt}	Cost of environmental externalities

x represents product or service, and t represents year.

These costs are essential to carry out project appraisal that deals with environmental as well as social sustainability (service delivery) in the short to medium run at least. However, some of these costs are difficult to quantify, especially the costs of environmental externalities. All the costs need to be standardised by annualising the costs. Some of these costs like OpEx are incurred annually, while others need to be annualised. For these investments, past or future, we need to arrive at the present value of these investments in order to make the investments comparable across the schemes. Accordingly, Eq. 2.1 can be written as follows:

$$LCC_{xt} = f \left\{ \sum_{t=1}^n \text{pvf}_{xt} (\text{CapEx}_{hw_{xt}}; \text{CapEx}_{sw_{xt}}; \text{CapManEx}_{xt}; \text{CoCap}_{xt}; \text{DsCost}_{xt}; \text{IDsCost}_{xt}; \text{OpEx}_{xt}) \text{CoEExt}_{xt} \right\} \quad (2.2)$$

where

pvf Present value factor $(1 + r)^t$

r Rate of interest or inflator

t Time period

Rate of inflation or the prevailing rate of interest may be appropriate for estimating the present value or worth. Other alternatives include effective interest rate (rate of interest–inflation), national GDP inflator could also be used. Once the whole life costs are estimated, unit costs and annualised costs can be worked out.

2.5 Risk-Based Life-cycle Cost Analysis and Simulations

Some of LCCA components are characterised with risk and uncertainty. Systems fail randomly and may not follow a time schedule. Time required for rehabilitation/repair and costs may vary. As a result, while normative lifespan of different systems may not vary much, the actual lifespan varies due to risk and uncertainties associated with natural factors and unexpected climate events. The risk and uncertainty are often high in the case of products and services associated with natural resources. The risk factor can be modelled using probabilistic phenomena, that is by estimating the probability of risk in a particular location due to a particular event. In the event of risk, the earlier Eq. (2.2) could be written as follows:

$$LCC_{xt} = f \left\{ \sum_{t=1}^n \text{pvf}_{xt} (\text{CapExhw}_{xt}; \text{CapExsw}_{xt}; \text{CapManEx}_{xt}; \text{CoCap}_{xt}; \right. \\ \left. \text{DsCost}_{xt}; \text{IDsCost}_{xt}; \text{OpEx}_{xt}; \text{CoCEExt}_{xt}[\text{Psf}_{xt}] \right\} \quad (2.3)$$

where

Psf_{xt} Probability of risk

This formulation is more appropriate in the case of WASH services, as the dependence on groundwater is quite substantial. In this case, the total life-cycle cost is modelled as a random variable that is the sum of several cost items. Of these variables, the CapManEx is a random variable. The randomness or the probability of failure could be estimated using the observed values from the real-life costing in different agro-climatic locations. These observations can be complemented with expert opinions.

Risk and uncertainty analysis is often carried out using scenario building. Different scenarios are built using assumptions pertaining to the expected risks. Scenario building gives a band or range of possible options to choose from, and simulation models are used to arrive at scenarios. Monte Carlo simulation techniques are used to join probability distributions and economic data to solve

problems of uncertainty using spreadsheet techniques (Barringer and Weber 1996). Monte Carlo simulation techniques use random numbers to generate failure data and cost data considering the statistical distributions. Monte Carlo results are similar to real life because the results have variations around a given theme.

Here is how the Weibull database and Monte Carlo simulations work using the coupling data as an example. Given $b = 2.0$, and $h = 75,000$ h, what is a Monte Carlo age to failure? Solving the Weibull equation for time,

$$t = h * \{\ln(1/(1-\text{CDF}/P))\}^{1/b}$$

where CDF is the cumulative distribution function or the probability of failure, which always varies between 0 and 1. The CDF/P range is convenient because spreadsheets also have a random number function, which varies between 0 and 1. This means if the $\text{CDF}/P = (\text{arrived}/\text{chosen by a number between 0 and 1}) = 0.3756$, then the Weibull age to failure is 51,470 h (or 5.9 years) as driven by the random choice of the number 0.3756. Contrast the Weibull results for age-to-failure with results from the exponential distribution, ($b = 1$) age-to-failure, which produces 35,322 h or 4.0 years using the same random number. When the random numbers are used repeatedly, then specific ages to failure are selected as representative of specific ages to failure. Alternatively, the probability of failure can be estimated using the historical data or expert opinion. Different random numbers or probability scenarios can be modified to build more complex failure propagation tables taking into account how good maintenance practices will reduce the number of failures occurring each period (Barringer and Weber 1996).

2.6 Methods and Tools of Environmental Impact Assessment

While all the relevant life-cycle costs are available in primary or secondary sources or derived from market prices, the cost of environmental impacts needs to be estimated. Various methods have been used to estimate the environmental impacts, as the environmental goods and services are not often available in the market. Here, we discuss some of the important methods used in estimating the impacts.

Methods¹ used in valuations of environmental impacts or costs and benefits can be broadly grouped as direct and indirect. Indirect methods² use actual choices made by consumers to develop models of choice for market and non-market

¹ We restricted to the methods appropriate for this section. We have not dealt with financial or economic appraisal methods such as Cost-Benefit Analysis (CBA), Cost-effectiveness Analysis (CEA), Multi-Criteria Analysis (MCA), Risk-Benefit Analysis (RBA), Decision Analysis (DA), etc.

² These are also known as surrogate market valuation approaches, when information about a marketed good is used to infer the value of a related non-market good.

goods. These include most importantly human capital (HC) approach, replacement cost method, travel cost method (TCM), hedonic pricing method and loss of production method. Direct methods ask consumers their maximum willingness to pay towards a possible change (improvement) in environmental amenities. These methods fall under stated preference techniques where individuals do not make any behavioural changes but state how they would be behaving. The direct methods include contingent valuation method (CVM) and contingent ranking or contingent behaviour. Here, we discuss the methods that are used in assessing the environmental impacts.

2.6.1 Indirect Methods

HC method is most widely used in estimating health-related environmental impacts. The HC approach considers people as economic capital and their earnings as return to investment. Environmental economics focuses on the impact on human health due to bad environmental conditions, and the effect this has on the individuals and society's productive potential. HC approach provides an estimate of direct and indirect burden resulting from the prevalence of disease during a given period. Prevalence of disease-based estimates and present value of future costs are calculated. In the case of incidence of disease, the present value of future direct costs (mortality) and indirect costs morbidity ought to be calculated. There are also non-health sector costs, which are often difficult to estimate due to data limitations. Non-health sector costs include psychological costs such as the influence of mortality on family, life cycles, divorce, widowhood, orphan hood, etc.

Direct health costs are the costs incurred due to mortality and morbidity. Indirect health costs are the value of output lost because of loss of productivity in terms of working or keeping house. Here, the method would estimate the economic costs of illness of a productive human being. Two variants of this can be taken into account while measuring economic costs of illness due to environmental factors: first the cost of medical treatment and second the loss of earnings (working days) due to illness. Together, they provide the total economic loss due to ill health. However, it may be noted that these estimates need to be corroborated with medical science or epidemiological data to correlate the illness with pollution. One way is to conduct laboratory tests of various water samples from the sites in order to check the presence of water-related diseases. The linkages between water pollutants like arsenic and other metals and health hazards are well established; discussions with local doctors help in establishing the linkages between water pollution and the prevailing diseases in the locations. The HC approach provides valuable information, provided its limitations (especially information) are addressed. Though it cannot provide an accurate or complete estimate on the value of life, it does indicate economic costs due to morbidity and premature mortality.

In the context of poor water quality, households adopt various mechanisms. Some buy water, some travel farther to fetch good-quality water, and some knowingly (due to lack of affordability) or unknowingly consume poor-quality water. The last category of households might incur other costs such as medical treatment, etc. In a case study in Andhra Pradesh, India, about 5 % of the households still drink sewerage-contaminated water due to compulsions of non-affordability or non-availability of persons to bring water from nearby town (Kurian et al. 2008). Those who consume the water complain about stomach pain, diarrhoea and joint pains. Women complain that the water quality is getting worse over the years and they are now scared to use the water even for domestic uses. Families consuming this water may have to spend about \$5–\$8 per month towards doctor fees and medicines. However, there are no serious health complaints of severe sickness leading to loss of working days in the study region. The estimated total costs of water contamination come to \$88,763 per year for the entire village (Table 2.2).

The incidence of sickness and unable to work due to pollution was estimated to be between 48 and 50 days in another study in Andhra Pradesh, India, where water was polluted due to discharge of industrial effluents (Reddy and Behera 2006). The effective number of working days lost depends on the probability of getting employment. The average per-year-per-household loss of working days was calculated using the market wage rate in the villages. The estimated average loss per household due to loss of working days was about \$28. Number of visits to the doctors before pollution and after pollution and its expenditure revealed a substantial increase. Households in the region used to visit doctors 4–5 times in a year and spent \$3–4 on health, but after pollution, it has increased substantially in the affected villages, which has an adverse influence on the socio-economic conditions of the people in the affected villages. Expenditure on health depends on two factors: (a) the severity of diseases and (b) the economic condition of the family. Small and marginal farmers (owning to less than 2 ha of land) visit doctors 20 times per annum, and their expenditure on medical services is \$30. However, in

Table 2.2 Health costs of water pollution accruing to households (HH) using various methods in Bommakal Village, Andhra Pradesh, India

Indicator	No. of HH	Economic cost per household/year in US\$	Total cost in US\$/year
No. HH buying water (averting cost)	20	95	2,000
No. of HH fetching water from town (travel cost)	900	900 @ each HH spends an hour per day in fetching water and the wage rate is US\$0.30 per hour (total: \$270)	86,447
No. HH drinking contaminated water (human capital)	80	6 (medical expenses)	421
Total	1,000	196	88,868

Source Kurian et al. (2008)

case of medium (owning between 2 and 5 ha of land) and large (owning more than 5 ha of land) farmers, the average number of visits to doctors is 25 and 12 and the amount spent on medical expenses is about \$40 and \$60, respectively, after pollution. These differences can be attributed to two factors mentioned above.

In the case of China, HC approach was used in the case study of Chongqing region, China. Three components of cost of illness were considered, viz. medical treatment, loss of work and premature death. Three diseases were linked to contaminated water (i.e. hepatitis, dysentery and selected cancers) and were taken into account for estimating costs. In the case of premature death, loss of earnings during the working age (18–60) due to death was estimated. Median age of the patients to die is estimated as 53 years; hence, the cancer patient loses seven years of working life. Individual contribution to production is estimated using the per-capita growth (8 %) and a discount rate of 12 % were used. The total loss due to health damage was estimated to be \$21.7 million when HC approach was used (Yongguan et al. 2001).

Averting Costs (AC) approach states that in order to avoid the damage due to environmental degradation, one has to spend some money. For example, the victims of environmental damage replace their environment by moving away from the affected area. The costs, which the victims incur by moving to a clean/healthy environment, are called averting or replacement costs. One of the techniques adopted in the averting cost method is that of direct observation of actual spending on safeguards against environmental risks. In the context of health impacts, households may either treat the water on their own (filtering, boiling, etc.) or switch over to bottled water in order to avoid adverse health impacts due to drinking of unclean water. Data pertaining to the AC are based on the households' actual spending on treatment of drinking water and purchased bottled water from the market. In a study of Andhra Pradesh, India, it was observed that about 2 % of the households buy water from the market in order to avoid adverse health impacts. The estimated cost of this averting behaviour is estimated at \$95 per household per year (Table 2.2). In the arid regions of Rajasthan, almost a quarter of the total households buy water from the market and spend more than \$1 per day per household (Reddy 1999). Water treatment costs are estimated at about \$0.9 million in the Chongqing region of China.

TCM uses peoples' actual behaviour and hence captures the actual use values. Travel cost models are based on an extension of theory of consumer demand, with specific reference to value of time. This method, which is the most straightforward of the indirect methods, recognises that visitors to a recreation site pay an implicit price—the cost of travelling to it, including the opportunity costs of their time. Though this method is often used to estimate the willingness to pay for the facilities of a site using information on time people spent on getting to a site, a modified version of this method can be used to estimate the value of time. This method (random utility theory approach) is based on the assumption that the households' source choice decisions depend upon at least two sets of explanatory variables: (i) source attributes, which affect the households' utility, and (ii) household characteristics, which reflect difference in tastes and preferences. According to random

utility theory, the probability that household ‘*h*’ chooses alternative source ‘*j*’ equals the probability that the utility derived from using source ‘*j*’ is greater than the utility derived from any other alternative. Following this framework, utility function is estimated with two sets of variables, i.e. source attributes and household characteristics.

The functional form is as follows:

$U_{ih} = f[X_{ih}, Z_{ih}]$

where,

U_{ih} Utility derived by household ‘*h*’ using a source site ‘*i*’. Here, utility is indirectly determined by the choice of the source [site].

X_{ih} represents source attributes like distance between source and household, time spent, money paid for collecting water, etc.

Z_{ih} represents household characteristics like income, social status, education level, preferences, etc.

In this model, the dependent variable (source/site) is a dichotomous variable, and hence, it is estimated with the help of conditional logit model.³ This model is found to be useful in estimating the household’s value of time and hence suitable for adaptation in the context of valuation of resources. Two clear cases of such adaptation are drinking water and fuel wood where rural households spend substantial amounts of time in fetching/hauling them. In this (conditional logit) model, the value of time spent by the household is given by the ratio of the two coefficients measuring time and money spent for water or fuel wood by the household. Here, the value of time is defined as the marginal rate of substitution between the time spent in collecting water/fuel wood and money paid⁴ for them. Health costs of using poor-quality water can be estimated if households have access to two sources with different source characteristics in terms of quality, time spent/money spent. For instance, the extra effort put in/amount spent by households for collecting/buying good-quality water is the value households place on health. One problem that may arise here is the existence of markets for these items. It may be difficult to find markets for drinking water and fuel wood in all the regions, especially drinking water. Another problem⁵ here may be the large variations in tastes, availability of alternative sources, incomes, etc., which can be taken care of with appropriate econometric techniques. On the whole, TCM is believed to be a useful tool and found to have worked well in different contexts.

³ Conditional logit is used to deal with the data structure, which includes both groups of independent variables—source attributes vary across sources while household characteristics do not vary across sources.

⁴ This is calculated in terms of the price times the quantity of water/fuel wood consumed per day. In other words, the values of water/fuel wood if purchased at market price (even if the household is not actually purchasing them in the market).

⁵ Various other problems related to Travel Cost Method are not considered here.

Greater proportion of households resort to fetching water from far-off places to avoid the ill effects of poor-quality water, which is mainly due to their poor economic status in the developing countries. Villagers go to nearby towns to fetch water from municipal supplies. Studies observed that households spend an hour to bring two cans of water (about 40–50 l) and in summers, it becomes worse as wait times can last 2–3 h (Kurian et al. 2008). About 80 % of the households resort to this mode, and the estimated costs are \$270 per household per year. In arid regions, the travel time tends to be substantial, i.e. 18 h per day per household. Often, these travel costs are not accounted as the opportunity costs of labour, especially for women and children (the main fetchers of water), tend to be zero in some rural areas of developing countries (Reddy 1999).

Hedonic pricing method uses surrogate markets to impute values of non-market goods. It estimates the implicit price of the non-market characteristics, which differentiate closely related or explicitly similar products. This method is widely used to value environmental amenities or disamenities associated with a good, using market values (i.e. property valuation approach or wage differential approach). For instance, take two units of houses, which are identical in all respects except one, i.e. air pollution. Their prices would differ if people place value on clean air or the health. If so, the difference in market price between the two units should, *ceteris paribus*, reflect the willingness to pay for better air quality/healthy environment. Similarly, wage differentials in similar jobs can be attributed to working and living conditions. In other words, a higher wage is needed to attract workers to polluted environments or unhealthy industries like coal mining, nuclear complexes, etc.

Like in the TCM, here also people's willingness to pay for healthy environment can be arrived at by estimating a regression equation and then deriving the demand function. The functional form of the equation may be specified according to the good that is being valued. In the present context, hedonic pricing method is appropriate to derive the users' valuation of health in terms of clean air, availability of quality water, etc. Here, the functional form would be as follows:

$$PL_i = f[AQ_i, AW_i, QW_i, SP_i, OT_i]$$

where

PL_i Price of the property (house)

AQ_i Air quality

SQ_i Soil quality

AW_i Availability of water

QW_i Quality of water

SP_i Size of the property

OT_i Vector of other attributes like distance from the market, other amenities available, neighbourhood, etc.

i Index of properties ranging from 1 to n

One or more of these environmental qualities of the property (house) can be incorporated depending on the characteristics of the property. Hedonic pricing method is widely used with reference to air quality. The main problem associated

with this method is rooted in its stringent assumption of well-functioning markets. Given the variations in property prices, this may prove useful in deriving the users' willingness to pay for various environmental qualities and health. However, interpretational problems may arise if supply adjusts to changes in prices, which may result in insignificant or zero variation in prices (of property). But, this was not found to be a frequent problem as hedonic price technique is being used effectively and satisfactorily to estimate the impact of environmental factors on property values, though it showed poor results in the context of unknown or unclear (to the individuals affected) impacts. Though these methods have the potential to deal with the valuation of natural resources, in market as well as non-market situations, one has to see whether these estimates conform to local peoples' perspective. Utmost care is needed while interpreting the results, which ought to be complemented by qualitative information.

2.6.2 Direct Methods

The most important and widely used direct method is the CVM. Of late, researchers have also employed contingent ranking or contingent behaviour to estimate individuals' willingness to pay for environmental amenities. However, the development of CVM has been very rapid due to its extensive use and hence the problems and solutions associated with it. Despite numerous criticisms levelled against it, a reasonable degree of success and persistence led to increasing attention on CVM findings.

CVM is a modern name for survey methods. Only difference is that CVM elicits how people would respond to hypothetical changes in some environmental resources. CVM deploys direct valuation questions relating to individuals willingness to pay⁶ for certain environmental changes. These questions may be in the form of referendum or payment card apart from the direct questioning of the exact amount an individual/household is willing to pay (WTP). However, the direct questioning has been criticised as a difficult question to answer. The referendum approach includes dichotomous-choice, close-ended, or take-it or leave-it question formats, while the payment card format specifies a range of values from which the respondent is asked to mark the highest value he or she would be WTP. Another way of eliciting information is through a bidding game procedure, which is somewhat similar to payment card approach, where the respondent is offered different hypothetical bids until a range is generated. In this, the true willingness to pay is expected to lie between positive and negative responses rather than on a single point.

A general criticism of CVM is regarding the validity of insights derived from people's responses to hypothetical situations. How far out are the estimates

⁶ The concept of willingness to accept is less preferred as it is observed not to reflect the true picture—often found to be giving over estimates when compared to willingness to pay estimates.

reliable or accurate? Awareness of the respondent with regard to the suggested environmental amenities, which are often esoteric (like polar bears, acid rain, rain-forests, etc.), is critical for obtaining reliable estimates from CVM. Lack of knowledge regarding the ‘good’ or ‘bad’ in question may result in hypothetical answers as well. Due to this reason, CVM is the most scrutinised among the social sciences research methods, to our knowledge. However, when questions are asked regarding the issues closely related to the respondents such as health, clean water, etc., CVM is found to provide results that are more accurate. This, however, would limit the types of commodities or decisions that can be included in CVM analysis. CVM is observed to provide theoretically consistent and plausible measures of individual values for some types of environmental resources.

CVM is best suited in the event of hypothetical or missing market situations. CVM can generate reliable estimates of willingness to pay even for ‘goods’ or amenities in a market situation. Here, one is often talking about the improvements in the present situation rather than a status quo position. If CVM is credible in estimating non-market values, it should be at least reliable in market situations. Nevertheless, due caution has to be taken to avoid creeping in some of the important biases while using CVM. These biases include (i) sampling bias, (ii) non-response bias, (iii) strategic bias, (iv) hypothetical bias, (v) part-whole bias, (vi) information bias, (vii) aggregation bias, (viii) interviewer and respondent bias, (ix) payment vehicle bias and (x) starting point bias. Some of these biases are specific to CVM, while others are endemic to all survey methods. CVM surveys can be designed to reduce the bias problem to an acceptable level such that undertaking a CVM evaluation does provide us with useful value estimation information.

Given the poor quality of drinking water, households are WTP for improved water quality. In a study of six villages, not a single household expressed ‘no’ to the WTP question for the provision of quality water by a private firm (Reddy et al. 2009). This is true for both capital costs and membership fees, which is fixed at a nominal level and user charges. Majority of the households are WTP \$0.75 and more as membership fee for safe drinking water. Majority of the non-poor households are WTP \$1 and more (Table 2.3). In the case of user charges, all the households are WTP the present rate of \$0.038 per 12 l. Most of the households prefer home delivery of water, and they are WTP extra for the transport. Among the non-poor households, 82 % are WTP extra (i.e. \$0.063/12 l). In the case of

Table 2.3 Willingness to pay water (per can of 20 l) (per cent households)

Costs in \$ per 12 l	Poor	Non-poor
<i>Capital costs</i>		
Up to \$0.75	80	40
More than \$1	20	60
<i>User charges</i>		
User charges as per cent to income	4.8	0.93
User charges as per cent to expenditure	4.5	3.5

Source Reddy et al. (2009)

poor households, only 52 % are WTP this price. The differences between poor and non-poor households in the WTP bids are due to the differences in ability to pay. Ability to pay is examined by looking at the household income and expenditure figures. As a proportion of household incomes, poor households are WTP 4.8 % against 0.93 % in the case of non-poor households. This is quite substantial by any standard, as it is often assumed that households are WTP up to 3 % of their income. In terms of expenditure, poor households are WTP more than their counterparts (Table 2.3). It is also observed that willingness to pay for public provision of drinking water is often lower when compared to private supplies. This is mainly due to lack of trust in public utilities in the developing countries.

It is observed that the willingness to pay estimates is often on the higher side when compared to HC approach. A comparison of these two methods in the Chongqing region of China revealed that WTP estimates are more than three times higher than that of the estimates from HC approach. A major share of this goes to the avoidance of premature cancer deaths due to water pollution (Yongguan et al. 2001). The difference in estimates could be due to the reason that WTP estimates often include non-tangible/non-economic costs, viz. social, psychological, aesthetic values, etc.

Contingent ranking or contingent behaviour is an indirect approach within the direct methods. In contingent ranking, respondents are asked to rank some non-market resources in the order of their preference. These non-monetary preferences are then 'anchored' by simultaneously asking respondents about some of the familiar items like hand pump or bore well. Then, respondents are asked for their willingness to pay (WTP) for the familiar items. These WTP estimates are then used to infer the WTP for non-market sources. Though this method appears to be simple and capable of generating reliable WTP estimates, it may not provide the true estimates of WTP apart from having additional biases, which are not identified so far. Another pertinent problem may arise due to the difference in ranking non-monetary priorities of individuals and their valuation in monetary terms. This mismatching of monetary and non-monetary priorities is found to be substantial in the context of local-level valuation of resources.

In the case of contingent behaviour approach, respondents are provided with alternative scenarios of environmental amenities from which they have to make a choice. This facilitates the explanation for the choice of one alternative over others as a function of attributes, which include travel distance, etc. This method makes data generation more complex, as it tends to confuse respondents by giving them multiple choice of amenities. This method also involves more sophisticated econometric techniques for estimation purposes. More importantly, this method when combined (jointly estimated) with revealed preference (indirect) approach is expected to provide a fruitful variant to CVM rather than by itself.

Keywords and Definitions

AC	Averting costs
CBA	Cost–benefit analysis
CVM	Contingent valuation method
DA	Decision analysis
HC	Human capital
Life-cycle	‘Consecutive and interlinked stages of a product system, from raw material acquisition or generation from natural resources to final disposal’ (ISO 2006)
Life-cycle approaches	‘Techniques and tools to inventory and assess the impacts along the life cycle of products’
Life-cycle assessment (LCA)	‘Compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout life-cycle costing’ (LCC): ‘Life-cycle costing, or LCC, is a compilation and assessment of all costs related to a product, over its entire life cycle, from production to use, maintenance and disposal’ (UNEP/SETAC 2009)
MCA	Multi-criteria analysis
RBA	Risk–benefit analysis
TCM	Travel cost method
WTP	Willingness to pay

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