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Systems performance analysis of Oslo's water and wastewater system

Thesis for the degree of Philosophiae Doctor

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Abstract

Introduction: An advanced urban water and wastewater network – from the source of raw water to the sink for the treated effluent wastewater – is, to say the least, a complex one. The interdependencies and interrelationships among the constituent network components make an integrated network analysis as necessary for an as-thorough-as-possible understanding of the system, as a separate analysis of each of the different network components. If sustainable development is to be pursued by urban water and wastewater utilities, a foreknowledge of the evolution of the network to its configuration at the time of the analysis, is a *sine qua non.* In simple terms, what is observed now, is the result of all that has been done in the past. More specifically, this evolution over time, has called for, and has been associated with, material inflows and outflows, energy consumption and related emissions, environmental impacts along the way, periodic capital investments to extend, expand and upgrade the systems, annual expenses on operation and maintenance, and changes in policies, rules and regulations at the administrative level.

Materials and chemicals, energy and money, in addition to time and manual labour, are the 'factors of production' employed to fulfil the twin goals of water supply and wastewater treatment. The anthropogenic network components managed and operated by the utilities, are the water treatment plants (WTPs) and wastewater treatment plants (WWTPs), water pipelines, sewage, storm-water and combined-flow pipelines, and water and sewage pumping stations. (It goes without saying that the consumers 'mid-stream' linking the water supply sub-system to the wastewater handling sub-system, constitute the *raison d'être* of the network). Utilities should aim at providing acceptable levels of service to the consumers, while optimising the expenditure of money, the consumption of energy, chemicals and materials, and reducing environmental impacts. This is the triple bottom line approach (social-economic-environmental) which needs to be incorporated into asset management of the 21st century.

Background of Oslo: The city of Oslo – the focus of this research – is inhabited by about 600,000 people (as in year-2010); and is serviced by three WTPs of different capacities - Oset, Skullerud and Langlia - drawing raw water from the lakes Maridalsvannet, Elvåga, and Langlivannet, respectively. The treated water from the three WTPs reaches the consumers in the domestic, industrial and commercial sectors of the city through approximately 35,000 water pipes with a total length of over 1,500 kilometres. The sewage discharged by the consumers and the storm-water (rainwater and snowmelt) are transported to two WWTPs – BEVAS (Bekkelaget Vann AS) and VEAS (Vestfjorden Avløpselskap) – through more than 54,000 pipes with a total length of around 2,200 kilometres. Water and sewage pumping stations pressurise the respective flows. The treated effluent wends its way into the Oslo fjord, which is contiguous with the Atlantic Ocean.

IE tools and methods: The longest time-span considered for the time series analysis is 16 years – for the water and wastewater pipelines. For WTPs and WWTPs, the time window is much shorter - from year-2000 onwards. Material flow analysis (MFA) is performed to study the inflows of pipeline materials into the water and wastewater pipeline networks in Oslo. The phenomenon of pipeline stock saturation is discussed *vis-à-vis* two other Norwegian cities – Trondheim and Tromsø; and an embodied energy analysis (EEA) is performed. Environmental life-cycle assessment (LCA) is carried out with the results of the MFA serving as the platform, to translate the past annual inflows into their associated environmental impacts, and to forecast the impacts that would occur in the future. Life-cycle costing (LCC) is performed in order to emphasize the importance of future investment decisions in, and rehabilitation approaches to the wastewater pipeline network. The flows of, expenses on, and the impacts associated with, chemicals and energy consumption at the WTPs and WWTPs, are analysed as time series. Energy, environmental and economic analyses are performed for the water and sewage pumping stations. Based on the sub-system studies, the system is visualised as a whole, and comparisons among the sub-systems are done. The elaborateness of the studies, when it comes to historical (time-series in other words) analyses, is limited only by the non-availability of detailed data, and the aversion to make too many assumptions.

Measuring sustainable development: Indicators are useful as metrics in order to measure a water-wastewater utility's progress towards sustainability. Sustainability or sustainable development, when considered holistically with regard to the urban water and wastewater system, may be looked upon as four-pronged. Social, economic, environmental and functional indicators can be aggregated by using suitable weighting factors to arrive at criteria indices and a grand sustainability index. Time series analyses like the ones referred to in the earlier paragraph will yield indicators as a time series, and enable a systematic measurement of 'sustainable development'. Targets and benchmarks can be set in order to stimulate progress. There are benefits and pitfalls associated with such an aggregation.

Key findings: Useful insights are obtained from the analyses referred to, in the earlier paragraphs. As the water and wastewater pipeline networks evolve towards saturation, the annual environmental impacts decrease over time, and are increasingly dominated by the operation, maintenance and rehabilitation phases. Concrete is the dominant pipe-fabrication material in the wastewater pipeline network, while ferrous metals dominate the water pipeline network. LCC enables one to prove the superiority of a physical lifetime approach over the in-vogue economic lifetime approach, when it comes to economising and managing/utilising the pipeline assets more efficiently. The comparison among Trondheim, Oslo and Tromsø yields an interesting correlation between the population density and the mass of pipeline materials per capita of the population, which needs to be confirmed by obtaining more datasets – from cities within Norway firstly and foreign cities thereafter.

The economic and environmental analyses of WTPs and WWTPs in the city give interesting results, when the energy consumption, costs and associated environmental impacts, expressed in terms of per-unit-service-delivered – unit volume of water supplied in the case of water treatment and unit volume of wastewater treated in the case of wastewater treatment – are compared with the corresponding values for chemicals. Eutrophication emerges as the dominant environmental impact when wastewater treatment and effluent discharge are considered, pointing to the possibility of channelling funds towards nutrient removal in the WWTPs, or looking upstream to initiate source control measures to impede the release of nitrogen and phosphorus into the wastewater. The capture and utilisation of biogas has played a significant role in avoiding the production of natural gas and electricity, and the associated environmental impacts.

Gleanings: Thinking of the urban water and wastewater system as a single entity composed of interrelated components may possibly be easier on paper, but translating the knowledge of the interconnectedness to the adoption of new approaches to the management of the assets, is beset with numerous challenges. In a complex system in which there are 'wheels within wheels', changes or modifications made in one part, may have immediate or delayed effects on the others. Just as the component parts of the system are interconnected, so are the social, economic, environmental and functional aspects of sustainability. The priorities are never the same over time. There are innumerable external factors beyond the control of the utilities – prices of energy and chemicals for instance – which need to be taken into consideration. Sustainable development of urban water and wastewater systems is verily a tight-rope walk. Sustainability studies are *never* completed. This one is no exception. There are numerous aspects which have not been integrated into the research, owing to time constraints, paucity of data, and the subsequent need for narrowing down the scope. This study would however form the bedrock for consolidations, extensions and forays into more detailed examinations of the system.

Acknowledgements

When one works for four years in a university setting for one's PhD, there are numerous positive influences which keep one moving towards the coveted goal. Every individual one interacts with and every place one visits constitute the chain of events leading to the completion of the thesis. And, of course, there are influences of the past – those words of encouragement, those friendly pats on the back, those prayers for one's well-being and success, and those expressions of trust and faith in one's abilities. Thanks are due to hundreds; and names left out, needless to mention, are not tantamount to gratitude not shown.

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This however, is just a new beginning…..in life, in work, and in the 21st-century world which has welcomed industrial ecologists with open arms. There is a lot more to do, from here onwards, in the company of people committed to the cause of sustainable development…

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R. Ugarelli, G. Venkatesh, H. Brattebø and S. Sægrov. Importance of investment decisions and rehabilitation approaches in an ageing wastewater pipeline network. A case study of Oslo (Norway). *Water Science and Technology.* 58(12): 2279-2293, 2008.

Paper no. 2: ([121] in List of references)
G. Venkatesh, J. Hammervold and H. Brattebø. Combined MFA-LCA of wastewater pipeline networks – Case study of Oslo (Norway). *Journal of Industrial Ecology*. 13(4):532-550, 2009.

Paper no. 3: ([123] in List of references)
G. Venkatesh and H. Brattebø. Environmental impact analysis of chemicals and energy consumption in water treatment plants: Case study of Oslo, Norway. *Water Science and Technology-Water Supply*. Accepted for publication on 31st July 2010.

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R. Ugarelli, G. Venkatesh, H. Brattebø and S. Sægrov. Historical analysis of blockages in wastewater pipelines in Oslo and diagnosis of causative pipeline characteristics. *Urban Water*. 7(6):335-343, 2010.

Paper no. 6: ([126] in List of references)
G. Venkatesh and H. Brattebø. Energy consumption, costs and environmental impacts for urban water cycle services: Case study of Oslo (Norway). Energy. 36(2):792-800. 2011.

Paper no. 7: ([127] in List of references)
G. Venkatesh and H. Brattebø. Analysis of chemicals and energy consumption in water and wastewater treatment, as cost components: Case study of Oslo, Norway. Under review with *Urban Water,* 2010.

Paper no. 8: ([130] in List of references)
R. Ugarelli, G. Venkatesh, H. Brattebø and S. Sægrov. Asset management for urban wastewater pipeline networks. *Journal of Infrastructure Systems.* 16(2): 112-121, 2010.

Paper no. 9: ([131] in List of references)
G. Venkatesh and R. Ugarelli. 'Oslo consumers willing to pay more for improved services.' Interview with Per Kristiansen [16]. J*ournal of the American Water Works Association.* 102(11):26-29, 2010.

1. Introduction

This introductory chapter is split up into two sections. The **Background** section dwells on sustainable development in general, describes it in the context of urban water and wastewater systems, and provides a background of the system in the city of Oslo. The **Motivation and Research Objectives** section outlines the purpose and scope of this research. (It must be mentioned at the outset that *British English* has been used as the language of the thesis.)

1.1. Background

1.1.1. Sustainable development

The term 'sustainable development' has come a long way since the Brundtland Commission defined it in the publication *Our Common Future*, in 1987, as 'development that meets the needs of the present without compromising the ability of future generations to meet their own needs' (World Council for Environment and Development [1]). From mere moralistic jargon in the late 1980s and early 1990s, it has evolved over the years, and now, connotes a set of practicable strategies which when implemented, would take one towards the condition or state labelled as sustainability. This state is known to be a moving target, and needs to be relentlessly pursued anew, every time the factors influencing it keep changing. One of the many publications in which this distinction between 'sustainable development' and 'sustainability' has been brought out clearly, is European Communities [2]. We thus have an elusive, impermanent end-goal which is pursued with a changeable set of means and ways. It has been stated in Quental, Lourenco and da Silva [3] that the introduction of sustainable development as a concept was an intellectual answer to reconcile the conflicting goals of environmental protection and economic growth.

Businesses incorporate sustainable development into their operations and rechristen it as a triple bottom-line or the 'triple P' – social (people), economic (profit) and environment (planet) – management approach, a term coined originally for the petroleum-major Shell in 1994 by John Elkington, who has referred to it as a 'win-win-win' situation in Elkington [4]. *Figure 1.1* depicts the foregoing discussion graphically. Far-reaching and all-encompassing social welfare, resource conservation and environmental upkeep should not be sacrificed at the altar of a blind pursuit of economic growth. The intra-goal conflicts indicated by the double-headed arrows linking the spheres should be eliminated in the march towards the elusive goal of sustainability.

Growth, development and progress are not interchangeable synonyms (Cameron and Neal [5].) It can be said that in order to sustain economic growth, there needs to be economic development (diversification of activity). In order to sustain economic development, there should be progress. Progress here entails the holistic outlook indicated by the triple bottom line approach. The key word here is 'to sustain' – to maintain for long periods of time. It follows thereby that for the sustenance of economic development, it is imperative that the social and environmental aspects are not overlooked.

When one talks about sustainability, one is usually expressing a desire to maintain some emergent property over long periods of time (Ehrenfeld [6]). The referred-to paper calls it a 'meta-quality'. What is desirable, according to Guha [7], is 'orderly growth; not growth at the expense of order or for that matter, order at the expense of growth.' The abstractness associated with it can be concretised to some extent by identifying and defining certain indicators – by following the processes of conceptualisation and operationalisation commonly used in the social sciences (Singhirunnusorn and Sternstrom [8]).

It was way back in 1992, at the Rio Earth Summit, that an emphasis was laid on the development of suitable indicators for the measurement of sustainable development, as aids for decision-making at all levels. Several papers have been published since then, on the fundamentals of sustainable development indicators, methodologies to identify, define and measure indicators and also the limitations thereof. An indicator, as an Organisation for Economic Co-operation and Development (OECD) report defined it, is a value derived from parameters, which points to, provides information about, and describes the state of a phenomenon/environment/area, with a significance extending beyond that directly associated with the parameter value (Keirstead and Leach [9]).

To understand the *status quo* that prevails at the time of writing, one needs to relate it to the past, because what obtains now is the sum total of all that has been, and occurred, in the past. It is here that a time-series analysis – a peep into history so to say – becomes important, when one wishes to develop sustainably into the future. Having seen and understood the present with respect to the past, the future course of action (continuous course corrections in other words) can be planned. As Cameron and Neal [5] have pointed out, 'A correct diagnosis of the origins of a problem does not in itself guarantee an effective prescription, but without such a diagnosis one can scarcely hope to remedy the problem.' If the world has to develop in a sustainable fashion, all the economies in it have to place sustainable development at the top of their respective agendas. For a national economy to advance towards sustainability, all the sectors comprising it need to embrace the triple bottom line approach. The urban water and wastewater system – a key component of urban infrastructures - with its complex forward and backward linkages to various sectors of the global economy, is one of them.

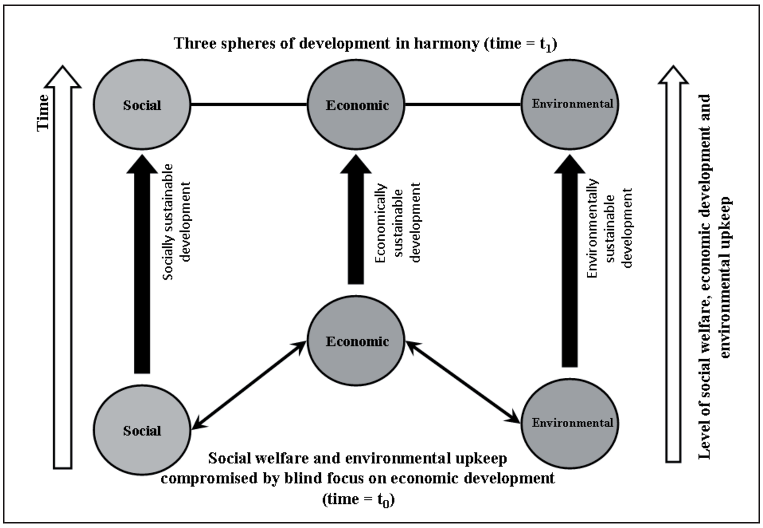

Figure 1.1: Sustainable development and the triple bottom line

1.1.2. Sustainable development of the urban water system

If water and wastewater utilities need to put sustainable development on their agendas, first and foremost, traditional urban water management which separates the systems of water supply, sewerage and drainage needs an urgent overhaul. It has to metamorphose into a more integrated system of management considering a complex array of values and factors – environmental integrity, social equity, landscape aesthetics, economic efficiency, integration of different professions and community engagement (van de Meene, Brown and Farrelly [10]).

The underlying *mantra* would be to provide the level of service desired by consumers and stipulated by regulations (value for the consumers' money in other words), while keeping a tight rein on the total expenses, optimising the consumption of materials, chemicals and energy, and progressively reducing - as and when possible - the environmental footprint of the water and wastewater system. It is this *mantra* which guides the decision-makers when strategies are drawn up. The urban water and wastewater system can be looked upon as a service niche in which sustainability indicators can be tested before transiting to urban sustainability as a whole (Keirstead and Leach [9]). In Alegre et al [11], six classes of performance indicators for water supply (water resources, personnel, service-quality-based, operational, economic & financial, physical) and five for wastewater treatment (environmental, physical, operational, service-quality-based, economic and financial) have been identified. Values for 14 different indicators (belonging to 12 defined indicator categories) for most of the countries of the world, have been published in The International Benchmarking Network for Water and Sanitation Utilities [12]. These include, *inter alia,* water and sewerage coverage, consumption, degree of metering, operation costs, coverage of operation costs and employment generation potential of the utilities.

Based on Hellstroem, Jeppsson and Kaermann [13], one can define four broad criteria – social, economic, environmental and functional. This, in essence, is the triple bottom line. The functional criterion is in the foreground - directly at the operational level – and is related to the other three which are in the background. Each of these criteria can be described and addressed (concretised) by a set of indicators. The number of indicators and the volumes of data to be handled will depend on how complex one wants the sustainability analysis to be. To identify and define the indicators, one can split up the urban water and wastewater system into its component parts (as depicted in *Figure 1.2* for instance, for the case of Oslo).

Every city will have some guiding criterion or a set of criteria under the sustainability principle. Under the criterion (or criteria), there would be some prime indicators. This will depend on what the pressing concerns and immediate challenges are. While the component-level indicators are essential for the lower-management and middle-management personnel who need to take concrete decisions regarding changes and improvements, the strategic management, bureaucrats and governments are concerned more about what Mitchell [14] has referred to, as the composite indicators (aggregations of the component-level indicators). The consumers also have an important role to play in the sustainable development of urban water and wastewater systems. According to Iyer [15], often, the demand side needs to be looked at carefully first before even thinking about the supply side. Even in cities where water scarcity or shortage is not a concern, wise and non-wasteful use of the resource is a must. Households, by using water-saving devices, and industries, by cascading or treating and reusing water, can play key roles in assisting the urban water and wastewater systems in their journeys towards sustainability. A gradual reduction in demand for treated water supply will lead to a decrease in consumption of, and expenditure on resources, and a control over negative environmental impacts.

Every water and wastewater system has its own unique organisation. Driven by governmental policies and regulations, demand from consumers, access to advanced technologies, and absence or availability of funds for capital investment, the system develops continuously. Changes are always imperative, just as the change to a triple bottom line management approach has become, of late.

1.1.3. Water and wastewater system in Oslo

Figure 1.2 depicts the water and wastewater system in Oslo, from source to sink. The numbers depicted are fair approximations of what prevails at the beginning of year-2010. All data in this sub-section have been sourced/derived/obtained/calculated from the following:

- Kristiansen [16]
- Brenden and Berger [17]

- Reksten [18]
- Toftdahl [19]
- BEVAS [20]
- Aasebø [21]
- VEAS [22]
- Selseth [23]
- Statistics Norway [24]

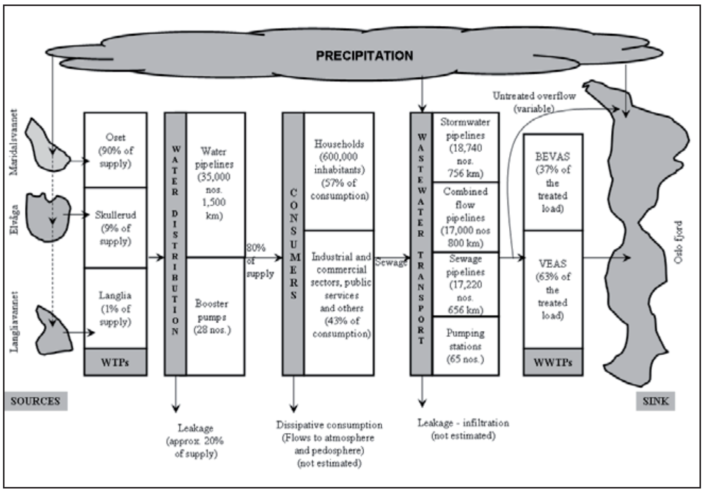

Figure 1.2: Depiction of the water and wastewater network in Oslo

The leakage from, and infiltration into, the wastewater transport network; and the dissipative consumption which does not enter the wastewater transport network have not been accurately estimated, and hence, are not indicated in the illustration. The untreated overflows which bypass the wastewater treatment plants (WWTPs) and run into the *fjord* are small *vis-à-vis* the flows entering the WWTPs. There is influent and effluent pumping at the entry and exit of the WTPs and WWTPs (not shown separately in *Figure 1.2*); and this is in addition to the pumping stations which comprise the water distribution and wastewater transport networks.

Figure 1.3 locates the WWTPs, the two major WTPs which account for 98% of the total water treated and supplied, and their respective water sources, with respect to the Oslo city centre and the Oslo *fjord.* The Oset WTP ('1' in *Figure 1.3)* which sources raw water from the Maridalsvannet lake ('A') accounts for 89% of the total water supply at the time of writing. The Skullerud WTP ('2') which draws water from the Elvåga lake ('B') supplies 9% of the total. The Langlia WTP which has not been depicted, accounts for close to 2%. The water distribution network is composed of 28 booster pumps and a network of water pipelines.

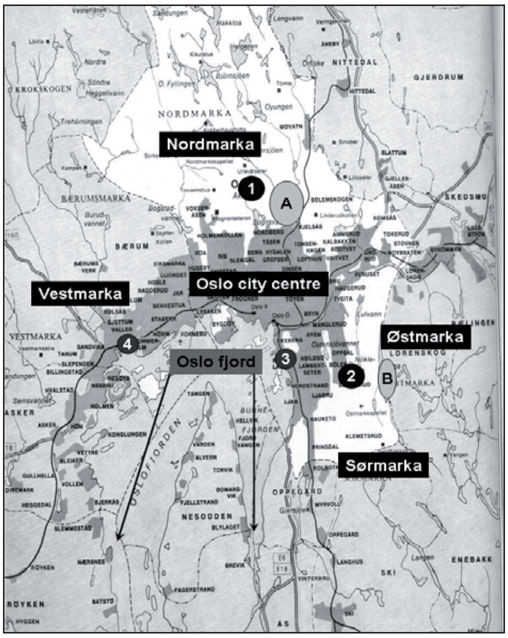

Figure 1.3: Geographical locations of the Oset (1) and Skullerud (2) WTPs, BEVAS (3) and VEAS (4) WWTPs, and the lakes – Maridalsvannet (A) and Elvåga (B); and the Oslo fjord.

On date, over 35,000 pipes totalling to a length of about 1500 kilometres and spread over 400 square kilometres, constitute the water pipeline network, which is almost saturated (i.e. not expanding much in size any more). Approximately 20% of the water treated and supplied by the WTPs to the distribution network is lost by way of leakage. Of the remaining 80% reaching the consumers, nearly 57% (45.6% of the total supply) is consumed in the households which account for a resident population of almost 600,000 at the time of writing. The remaining 43% (34.4% of the total supply) is consumed in the industrial units, commercial establishments and public services in the city.

All water consumed does not enter the wastewater pipeline network. There are the so-called consumptive flows – sub-surface seepage and evaporation, and incorporation of water into products in the industrial sectors. These have not been estimated. It follows that the volume of wastewater which exits the consumption phase into the wastewater pipeline network as sewage is less than the water which is consumed. However, in addition to sewage comes the stormwater from rainfall and melting of snow, and also infiltration of groundwater into the pipelines, giving a total wastewater flow which is higher than the water supply flow. The wastewater pipeline network can be trifurcated into stormwater pipelines, combined-flow pipelines and sewage carriers. These three categories, presently, account for approximately 34%, 36% and 30% of the total length of the wastewater pipeline network. About 2,200 km long, the network is almost saturated – additions to it have been few and far between, of late. At the time of writing, there are 65 pumping stations across the city, which pressurise the sewage *en route* to the WWTPs. In addition, there are 45,000 manholes, 528 stormwater outlets, 49 measurement stations, a septic disposal station, 2 retention pipes and 237 combined sewer overflows (as mentioned in Ugarelli et al [25]).

Calendar year	**Population of Oslo**	**Per-capita water supplied** (m^3 per capita p.a.)	**Per-capita wastewater treated** m^3 per capita p.a.)
2000	508,726	184.6	235.5
2001	512,589	182.0	215.3
2002	517,401	184.6	198.1
2003	521,866	177.8	202.4
2004	529,846	175.9	206.6
2005	538,411	174.8	206.9
2006	548,617	169.7	217.6
2007	560,849	169.6	198.7

Table 1.1: Changes over time, in Oslo's population and the per-capita values for water supplied by the WTPs and wastewater treated by the WWTPs.

Of the wastewater originating from Oslo and treated before being discharged into the Oslo fjord, 63% is handled by the Vestfjorden Avløpselskap (VEAS; '3' in *Figure 1.3)* and the remaining by the Bekkelaget Vann AS (BEVAS; '4' in *Figure 1.3*). The untreated overflows are small in comparison with the volumes being treated, as mentioned earlier. *Table 1.1* presents the changes in Oslo's population and the per-capita values for water supplied by the WTPs and wastewater treated by the WWTPs over time.

1.2. Motivation and research objectives

Infrastructure development is closely linked to economic development and social well-being. Ensuring that the two aforesaid goals are met, entails capital investments and operating / maintenance expenditures, consumption of materials and energy, and associated environmental impacts. Having developed different elements of the infrastructure in an urban setting, maintaining them is of paramount importance. Ageing assets deteriorate in efficiency and need to be overhauled and rehabilitated. Population growth complemented by a rise in the purchasing power, and thereby, an increase in demand for services, translates to greater stress on the infrastructure, accelerated wear and tear, capacity constraints, customer (or consumer) dissatisfaction, and a general reduction in welfare levels. The water and wastewater system in an urban setting fulfils basic needs – water supply and sanitation. The inhabitants have an inalienable right to demand a reliable supply of clean water and an efficient sewerage system. While compromising on the levels of the service(s) provided is out of question on ethical, humanitarian (and political) grounds, development should entail the optimisation of the consumption of resources – money, materials and energy - and the paring-down of the environmental footprint over time. The level of service, it must be mentioned, may increase in both quantity and quality, depending on consumer demands and government regulations. *Figure 1.4* illustrates the model framework on which the sustainability studies of the Oslo water and wastewater system in this research have been based *(Figure 1.2* is a more specific version, which depicts only the physical system). It depicts the physical system (the assets (A_i) in seven sub-systems) and the metabolism (water ($Q_{i,j}$), resources, wastes and emissions) associated with its functioning, along with the overarching influencing factors – economic, social and environmental – which determine its existence and progressive evolution, and the heads under which the overall system performance is measured.

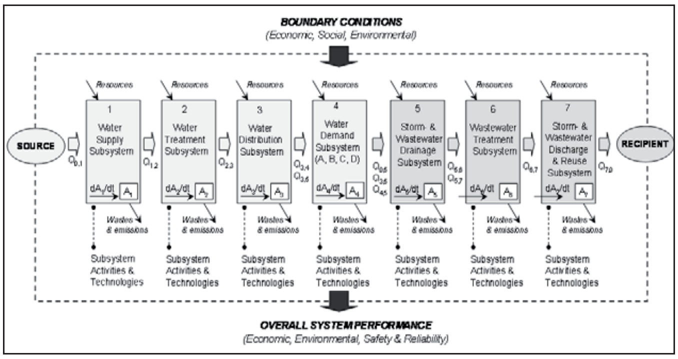

Figure 1.4: Metabolism in an urban water and wastewater system

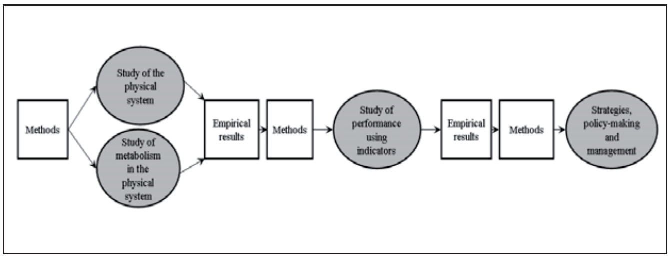

Figure 1.5: Envisioning the research goals and objectives

Talking of strategies, the utility – Oslo VAV (*Vann og Avløpsetaten)* - wishes to embark on the path towards sustainability. The *status quo* need not necessarily be the best-possible state, as the evolution over time may have been lop-sided. All the three spheres of development (*Figure 1.1)* may not have been on the agenda in the past. Knowing the 'wherefrom' is extremely important if one would wish to design strategies for sustainable development – the whereto, when and the how, so to say. *Figure 1.5*, which is self-explanatory, charts the outline of this research. Well-defined methodologies reinforced by industrial ecology tools and the empirical results - observations made by the application of the tools employed and methods devised – guide the research onward, towards a final practical usefulness. The understanding of the overall system performance in conjunction with a continuously-updated knowledge of the changing boundary conditions, enables the utilities to structure policies, and design effective strategies for the future. The grey ovals represent the broad objectives of this research.

A set of research questions are defined and structured according to the model in *Figure 1.5*. First, a group of questions dealing with theory and methodologies adopted in this research are defined, addressing the usefulness of the methods adopted for the study of the physical system, the metabolism in and the performance of this system, using indicators. Then, a group of questions dealing with empirical findings in this research are defined, addressing the characteristics of the system's performance, the reasons for the performance, and the possible future strategies for sustainable development, in policy-making and management for Oslo VAV in its capacity as the owner and operator of the system.

1. **Questions dealing with theory and methodology**

Material stock and flow analysis (MFA), energy analysis (EA) and environmental life-cycle assessment (LCA) are key industrial ecology tools which enable an understanding of the historical metabolism of urban water and wastewater systems; and also forecast the flows and environmental impacts for the future. Sustainability indicators are increasingly becoming popular as performance measurement tools for a variety of systems, but their usefulness to urban water systems has been of interest, only of late.

a) In what way does the use of the MFA, EA and LCA methods contribute to an in-depth understanding of an urban water system's physical composition, metabolism and environmental performance - today and over time?
b) How effective are these methods, and what are their limitations, as regards issues such as complexity of analysis, availability of data, and robustness of conclusions?
c) Is it possible to simplify the performance analysis of urban water systems, using selected sustainability indicators or indices, without compromising the system complexity?

2. **Questions dealing with empirical findings**

Over time, the challenges which utility managers encounter keep changing in form and degree of complexity. While ageing of assets is a prime concern, the environmental performance of urban water and wastewater systems has increasingly come under the scanner in the recent past. Asset management in the future entails taking the socio-cultural, environmental, economic, politico-legal and technological aspects into consideration.

a) What characterises the present sustainability performance of Oslo's urban water system - as a case study – with respect to the major performance challenges and the reasons for the same?
b) How do these challenges affect/influence the social, economic and environmental sustainability of the system? How can they be overcome?
c) How does performance change over time, and how is it linked to the system's physical state, ageing of assets, changing operation and rehabilitation practices, and corresponding changes in the metabolism of resource inflows?
d) Why is it important to adopt a systems approach as far as sustainable asset management in Oslo's urban water and wastewater system is concerned, and not look at the component sub-systems as 'islands of development'?

Answering these questions using the industrial ecology tools described in *Chapter 3* provides the 'bedrock' referred to in the *Abstract.* It also enables one to dwell on some 'what-ifs' for the future. These 'what-ifs' – numerous if one would list them as permutations and combinations – open up possibilities for furthering the research and translating the insights gained from the analyses to concrete action, by collaborating and working closely with the authorities at Oslo VAV.

2. Literature review

Callouts to 23 sources of reference (literature sources, personal communications etc.) appear in the previous chapter. A brief summary of the others is presented in this chapter. The literature sources have been split up into six categories for convenience. Some of these will be recalled in the subsequent chapters. The papers published by the author of this thesis in his capacity as the first author or co-author (those appended at the end of the thesis, and otherwise, except Ugarelli et al [25]) do not find mention in this one, as technically speaking, they do not comprise the *Literature Review per se*, but are rather products which have benefited from the same. Likewise, some other sources of information are referenced only in subsequent chapters.

2.1 Industrial ecology tools

Industrial ecology, as defined by Robert White (sourced from Ehrenfeld [26]) is *'The study of flows of materials and energy in industrial and consumer activities, of the effect of these flows on the environment, and of the influence of the economic, political, regulatory and social factors on the flow, use and transformation of resources. The objective of industrial ecology is to understand better how we can integrate environmental concerns into our economic activities. This integration, an ongoing process, is necessary if we are to address current and future environmental concerns.'* This systems approach of industrial ecology advocates thinking not merely of all the sub-systems as an integrated whole, but also of different aspects of the sub-systems. White's stress on the study of materials and energy and the effect of such flows on the environment, brings in Material Flow Analysis (MFA), Embodied Energy Analysis (EEA), Environmental Life-Cycle Assessment (LCA) and Life-Cycle Costing (LCC) as relevant and handy tools to be employed in sustainability studies. However, the study of flows and their impacts on the environment cannot be totally understood without factoring in several external factors (social, political, regulatory and economic as suggested by White in the definition above).

'Life-cycle analysis and mass balances (alternately known as material stock and flow analyses) will have a much greater effect on our daily life than we can imagine,' as said in a 1998 paper on urban water systems (Harremoes [28]). Brunner and Rechberger [29] have brought out the importance of MFA in urban metabolism studies and urban planning. In Binder [30], MFA has been combined with structural agent analysis in order to study the restrictive or enabling effect of social structures on material flow management. This is where the socio-psychological aspect kicks in. Knowledge of this aspect is crucial if sustainable consumption and pro-environmental behaviour are to be promoted (Stern [31] & Thøgersen [32]). In Brattebø, Bergsdal, Hammervold, Sandberg and Mueller [33], the authors present a model for material stock and flow analysis in the built environment, which can be applied to both historical analyses and forecasts for the future. Typifying the elements of the stock on the basis of different categories enables one, not just to understand the nature, causes and timings of the metabolism, but also to design effective measures to bring about changes and manage the same.

As far as EEA is concerned, in Ambrose, Salomonsson and Burn [34], a differentiation has been made between, from least to greatest in terms of magnitude, embodied energy associated with just the upstream manufacturing and production processes, embodied energy which takes into account the energy expended during the installation, and embodied energy associated with the life-cycle energy consumption. In Lenzen and Treloar [35] and in Treloar, Love and Holt [36], EEA has been applied to residential buildings – components of built urban infrastructure, which are serviced by the urban water and wastewater system. These, in other words, are life-cycle energy consumption analyses. Items numbered PRé Consultants [37], CML-University of Leiden [38], Swiss Centre for Life-cycle Inventories [39], Huijbregts et al [40] and Building and Fire Research Laboratory, National Institute of Standards and Testing, USA [41] pertain to the software, method, database, sources of normalisation and weighting factors, respectively, employed to carry out LCAs for this research. Rebitzer et al [42] provide a comprehensive and lucid background of LCA and also dwell on different methods, limitations and benefits thereof, and applications. The quality of the data used to perform LCAs is often beset by a range of uncertainties – reliability, completeness, and temporal, geographical and technological correlations (Weidema and Wesnæs [43]). This applies not just to the data used in LCAs but primary and secondary empirical data in general. The *modus operandi* of performing LCCs is explained in Fuller [44].

2.2 Asset management and pipelines

According to IAM/BSI [45], asset management can be defined as the set of systematic and coordinated activities and practices through which an organisation optimally manages its physical assets, and their associated performance, risks and expenditures over their respective lifecycles, for the purpose of achieving its organisational strategic plan. Asset management is a broad discipline which takes into its fold several methodologies ranging from knowing the assets better to structuring capital improvement plans (Buchanan [46]). Knowing the system better necessitates the maintenance of an information system (database) that would track assets and keep a tab on costs and reliability (American Society for Civil Engineers [47]). Such an information system, which records location, condition and criticality of assets (more relevant in the case of pipeline networks), enables effective asset management (Water Environment Research Foundation [48]). The usefulness of performance indicators – defined and measured by accessing the data from the said information system – has been discussed in Cardoso et al [49].

Within the urban water and wastewater system, the pipeline networks are the most expansive. It can also be said with a good degree of certainty that the oldest elements of the system's assets are to be found *sub-terra* - defunct and disconnected from the network, or functioning (post-rehabilitation) in concert with the other pipelines. In a saturated pipeline network like Oslo's, rehabilitation and repair dominate asset management practices. In the work of several authors (Cardoso, Coelho and Matos [50], Sægrov et al [51], Sægrov [52], [53] and [54], Fenner [55], Herz [56], Kleiner, Adams and Rogers [57] and [58], Sægrov et al [59], Shamir and Howard [60]), the importance of this phase in the life-cycle of water and wastewater pipelines has been highlighted, and related tools, methods and approaches have been discussed. In Mueller et al [61], cities have been referred to as 'mines of the future', thus making an indirect reference to, *inter alia,* the defunct ferrous pipelines which are no longer functional and are left beneath the ground – potentially recyclable entities. Embodied energy analysis (also referred to in Ambrose et al [34], Lenzen et al [35] and Treloar et al [36]) has been carried out in Ambrose and Burn [62] for pipe networks in Australia. According to the Concrete Pipeline Systems Association [63], in which the UK wastewater pipeline network has been studied, concrete emerges as the most environmentally-sound material for sewer networks.

2.3 Water and wastewater treatment

Ronald Droste's 'Theory and Practice of Water and Wastewater Treatment' [64] has been a veritable *vade mecum,* during the course of this research. Apart from a general technical understanding of water and wastewater treatment practices, the said book has also been a source of data for the analyses. The chemicals and energy aspects of water treatment have been examined by many researchers and several published works bear testimony to this. Racoviceanu et al [65], in a study of water treatment in Toronto, have calculated the contribution of the production and transport of treatment chemicals to energy consumption and greenhouse gas (GHG) emissions as 6% and 10% respectively. Among materials pressed into service in WTPs – piping, equipment, construction materials and chemicals, for a Californian case study performed in Stokes and Horvath [66], chemicals accounted for between 60% and 80 % of the total production energy consumption.

According to Vince et al [67], the production of carbon dioxide, lime, soda and coagulants accounted for over 50 per cent of the GHG emissions associated with the water treatment process life-cycle. The conflict with Racoviceanu et al [65] can be resolved when one considers the difference in electricity mixes (not explicitly stated though). In Travaglia [68], in a comparison performed between chlorine and hypochlorite as disinfectants in Australian WTPs, the authors have stated that sodium hypochlorite disinfection systems (storage, transport and usage) have been responsible for more accidents than chlorine gas systems and thereby challenged the widely-held notion that chlorine may be replaced by sodium hypochlorite to improve safety. Still on disinfection, Beavis and Lundie [69] have compared the use of chlorine (along with bisulphite), sodium hypochlorite and ultraviolet radiations as disinfectants from an environmental point of view and deemed the last of the trio to be the worst. Sodium hypochlorite, Beavis and Lundie [69] have claimed, outperforms chlorine, except when it comes to photochemical oxidation. Polyaluminium chloride, according to Zoubolis et al [70], is a more efficient coagulant than alum, and results in the production of treated water with lower turbidity and lower residual aluminium content. An LCA of water supply systems has been performed in Landu and Brent [71], using the Rosslyn Industrial Area in South Africa as a case study. According to Racoviceanu et al [65], the electricity use for water treatment is, in general less than that for water distribution and wastewater treatment. The specific energy consumption, as Stokes and Horvath [66] have calculated for the Californian case study, ranged from 17 MJ per m^3 for recycled water to 42 MJ per m^3 for desalinated ocean water with conventional treatment.

In Biehl and Inman [72], the authors, while citing a prior study carried out by the American Water Works Association Research Foundation in year-2008, have tabulated the operation and maintenance costs in a typical water treatment plant – salaries (35%), energy (34%), chemicals (16%), other materials (13%) and maintenance (15%). In wastewater treatment, as calculated in Tsagarakis et al [73], the expenses on chemicals accounted for between 4% and 8 % of the total operation and maintenance expenses in Greece – about one-tenth to one-fifth of the expenses on energy. In a study of Scandinavian WWTPs in Balmer [74], the corresponding value was 10%, while energy accounted for 25% of the total Operation and Maintenance (O&M) expenses. In the five Nordic plants studied in Balmer [74], the dependence on the external grid for electricity ranged from 6% to 100%. According to Clauson-Kass et al [75], electricity sourced from the grid by the Avedore WWTP in Denmark in 1998 contributed most to the global warming (8732 tonnes of CO_2-equivalents), while the energy derived from the biogas generated and consumed in-plant, was the largest contributor to acidification (44 tonnes of SO_2-equivalent, courtesy the sulphur dioxide emissions attributable to the combustion of the hydrogen sulphide in the 'sour' biogas). It is a known fact that there are significant losses of heat energy from WWTPs; and efforts can be made to recover this energy. The effluent from the WWTPs is a chief carrier of heat energy, just as the ambient air within the WWTPs and the sludge are. In Funamizu et al [76], the use of heat pumps to extract heat from the effluent wastewater in Japan has been discussed. Heated wastewater can also be very effectively used to warm flooring and melt ice in doorways, or even to heat the incoming fresh cold water. An emphasis has been placed on the need for harnessing the renewable energy potential of wastewater pollutants by resorting to anaerobic processes to generate biogas in Keller and Hartley [77]. However, Keller and Hartley [77] have also pointed out that the economics of energy recovery from sludge are governed by several constantly-varying factors. Nowak [78] has recommended a benchmark energy demand (excluding aeration) of nutrient removal WWTPs of 7-12 kWh (electrical) per person equivalent per year. While estimating costs for tertiary treatment of wastewater by rapid sand filtration with coagulants and ultraviolet disinfection, it has been concluded in Heinonen-Tanski et al [79] that the energy costs would account for 26 per cent of the operational expenses. Eastern Research Group [80] has stated that if all the 544 WWTPs in the USA which operate anaerobic digesters and have influent flow rates greater than 18,500 m^3 per day were to install combined heat and power (abbreviated in literature generally as CHP) units, 340 MW of clean electricity would be generated. This would offset over 2 million tonnes of carbon dioxide emissions annually. An interesting revelation has been made in Zhang et al [81], as a consequence of a comprehensive LCA analysis of a wastewater treatment and reuse project in China - the life-cycle benefit gained by treatment and reuse is greater than the life-cycle energy consumption for tertiary treatment, making the former approach feasible and favourable from an energy-saving point of view.

A Belgian case study – Lassaux et al [82] – has established that in WWTPs, the wastewater discharge into the final sink, dominates the environmental impact score calculated for wastewater treatment. However, unlike the observation borrowed from Racoviceanu et al [65] which appears earlier in this sub-section, this conclusion from Lassaux et al [82] cannot be considered to hold true *a priori* for all WWTPs – it would need to be tested piecemeal. Kawashima [83], while averring that infrastructure LCA is extremely important for policy planning, has also pointed out that over 85% of the life-cycle carbon dioxide emissions in activated sludge wastewater treatment occurs during the operation phase. Strategies to estimate, analyse and reduce GHG emissions from operations offer consumers solid evidence that their utilities are committed to positive stewardship of the environment, according to Strutt et al [84]. From Eckard [85], it is gleaned that in the USA, in the year 2006 alone, over 50 million tonnes of CO_2-equivalent GHGs were emitted from wastewater treatment, sludge handling and methane degassing.

Treatment of the wastes separated from the wastewater is as important as treating the wastewater to meet the standards of final discharge to the sink. In Soda et al [86], energy consumption and GHG emissions of different types of sewage sludge treatment systems in Japan have been analysed and the effectiveness of the use of digestion gas (biogas) for the energy needs in sewage sludge treatment processes, in reducing GHG emissions has been emphasized – akin to Keller and Hartley [77]. The feasibility of recovering phosphorus from wastewater plant sludge in Germany has been stressed on, in Cornel and Schaum [87]. The additional costs for phosphorus recovery have been put at €2 - €6 per capita per year in this paper. Grau [88] has stressed on 'affordability' and 'appropriateness' when it comes to developing urban wastewater treatment systems. In Rebitzer, Hunkeler and Jolliet [89], the authors posit life-cycle costing analysis (described in Fuller [44]) as a powerful method to expand the economic view while studying wastewater treatment.

2.4 Sustainability and the water and wastewater sector

Referring back to Robert White's definition of industrial ecology (Ehrenfeld [26]), it is understood that the objective of industrial ecology is to understand how to improve the sustainability of production and consumption systems, and implement solutions motivated by this understanding. The urban water and wastewater system is a complex production-consumption system to which industrial ecology tools can be applied. Water and wastewater systems in different cities face different challenges and so the strategies they chalk out for sustainable development, will differ from each other.

The technical and economic aspects of water and wastewater systems, needless to say, are strongly embedded in social and cultural dimensions, and therefore cannot be treated in isolation (Zerah [90]). Challenges due to development of urban centres – especially great urban agglomerations in developing countries are huge and water is a key figure in this equation, according to Varis and Somlyody [91]. The fact that the referred-to paper is over 12 years old at the time of writing indicates that researchers had started thinking on the lines of sustainability of urban water infrastructures in the last century itself.

When it comes to individual and governmental initiatives to conserve water and break the mould, so to say, Maher and Lustig [92] stress on the need for 'Water-Sensitive Urban Design'. In de Jong et al [93], it is stated that if solutions are to be found to overcome socio-political bottlenecks to sustainable development of urban water and wastewater systems, it is mandatory that a broad social awareness of the choices of the past and the necessity of radical changes in the future, should be created. Zerah [90] and Cheng [94] attribute the lack of seriousness as regards water conservation to the fact that water is quite cheap in many parts of the world (the said papers are case studies of India and Taiwan respectively). The strategies followed by managers and politicians in Spain have been mainly focused on the supply side of water management, whereas practices from the demand side have been systematically neglected over the years, according to Gascon et al [95]. The paper however also adds that both utilities and citizens have now realised that future trends in water supply will lie in water conservation programmes and retrofitting of facilities. Paul Reiter, the Executive Director of the International Water Association, observes in the June 2009 issue of the Water21 magazine that 70% of the energy consumption in the water cycle is tied up in the uses of water (mostly heating); and contends that if utilities would like to think of reducing energy usage in the water and sanitation networks (including the consumers), they need to focus on the customers. It has however been seen, as Nistor [96] writes about the situation in Moldova, that household consumption behaviour has been significantly affected by a rise in prices accompanied by the adoption of water meters. Between the years 1996 and 2006, the average per connection daily (pcd) consumption in Moldovan households dropped from a high of 328 litres pcd to 110 litres pcd. This shows that structural changes are often necessary in bringing out sweeping changes in consumer behaviour and enabling sustainable development of urban water and wastewater systems.

Fifteen environmental sustainability indicators have been defined in Lundin and Morrison [97]. The preoccupation with environmental impacts at the expense of economic and social issues, which has been plaguing several model-based assessments in the recent past, has been warned against, in Sahely and Kennedy [98]. Sustainability is not merely about the different aspects of the urban water and wastewater system, but also concerns the system's backward and forward linkages with the other sectors in the economy. Sustainability anywhere, according to Kissinger and Rees [99], is thereby linked directly and indirectly to sustainability everywhere. Yepes and Dianderas [100] have cautioned against the use of too many or too few sustainability indicators. The use of too many of them is likely to dilute the power of all of them, while the use of too few may not adequately describe the utility's performance and progress in reaching its goals. If the selection is done wisely and benchmarks are set, the indicators play a key role in enabling utilities to monitor their performance over time and design appropriate course-corrective strategies. When it comes to benchmarks however, Cabrera [101] has warned against inter-city comparisons, which in his opinion, would be akin to comparing apples with pears, if efforts are not made to understand the differences between the systems and the societies they serve.

Indicators can be defined, selected and measured over time, targets and benchmarks can be set, and the knowledge can metamorphose into corrective action. However, there are bound to be pioneers and laggards in keeping with the sigmoid diffusion curve for technology uptake (Zakkour et al [102]).

3. Research methods

All research begins with identifying the data needs dictated by its goal and scope. (The goal and scope, in this case, have been explained in the first chapter). Data gathering follows as a consequence. Availability and reliability of data may be questionable at times. Often, however, there is a sole authority to rely upon and thereby reliability may be less of a concern. If data have been recorded systematically and can be acquired with relative ease from the said source, this preliminary phase of the research can be concluded fairly quickly. But many a times, the need for maintaining and recording certain elements or components of data is not realised or appreciated by the said authority. There is a supply-demand mismatch, so to say, with the researchers demanding data which the owners (potential data suppliers) are not in a position to supply. While this certainly cannot be furnished as a pretext for stalling or not performing the intended research, one advances by seeking expert opinions, adopting proxies and making studied assumptions to fill the data gaps, while remembering that the scope for re-analysing always exists. The research will have to be revisited frequently, mended and modified, as and when non-available historical data become available.

It can be argued that sensitivity analysis makes more sense when one looks into the uncertain future and develops plans to manage assets, earmark expenditures and bring resources on-stream. Ideally, when everything is properly documented and recorded, there should be an acceptable low uncertainty about what has already happened. While the research results arrived at by taking recourse to assumptions, may lack accuracy, the ability of researchers to make the authorities realise the importance of monitoring, recording and maintaining data in a more comprehensive manner, is a positive fallout of this exercise. If sustainable development is on the agenda, the said imperativeness can be driven home more easily. This chapter discusses the data sources (including personal communications which enabled data acquisition and verification) and the application of some of the industrial ecology tools (described in subsequent paragraphs) to perform a historical analysis of, and some forecasts for, the sub-systems of Oslo's water and wastewater system.

3.1 Research methods

The methods adopted in this research can be broadly categorised into empirical and analytical.

The empirical methods can be split up further into the acquisition of primary empirical data – *document studies, database studies and personal communication with experts in fields related to the research* - and secondary empirical information – *literature surveys (already outlined in Chapter 2) and examination of methods and results from other research projects.* Data recorded by the utility – Oslo VAV – have been obtained from publicly-available databases, official documents and through personal communications with the chief officials at Oslo VAV. The personal communications included several meetings and interviews, taking the form of an iterative, closely- collaborative research process lasting for more than 2 years. The interviews referred to were semi-structured, varying in goal and scope. The ongoing research collaboration between Oslo VAV, SINTEF and NTNU facilitated access to competence and data at Oslo VAV. This was limited only by non-existence or non-availability of the data in the form requested. Literature surveys and examination of other research projects have provided insights into the results of studies carried out for urban water and sanitation systems in other parts of the world.

By virtue of its multidisciplinary nature, industrial ecology has a constantly-growing toolkit with research methods from diverse fields – the social sciences, economics, natural sciences and engineering sciences. The inputs to most of these methods are raw data – primary and secondary (derived or calculated from the primary, with or without the use of known numerical constants), which are gathered by following the afore-described empirical methods. The effectiveness of the analytical methods – the industrial ecology tools in other words - in the context of this research depends to a great extent on the outputs of the empirical methods. Most of the analytical methods are useful in performing historical analyses, understanding the present (*status quo*) and forecasting the future.

This research has adopted five of the industrial ecology tools to document different aspects of the system – Material stock and flow analysis (MFA), Energy analysis (EA), Life-cycle (environmental) assessment (LCA), Life-cycle costing (LCC) and Embodied energy analysis (EEA). However, as comprehensive data were not easily available when the research was being carried out, all these methods have not been applied to all the sub-systems. *Table 3.1* presents a brief overview of the scope and the extent of the application of the analytical tools. The insights gained by the adoption of these methods, however, mark a significant step forward in the sustainability studies of the system. In addition to the aforesaid industrial ecology tools, pipeline blockage analysis has been carried out for the wastewater pipeline network in Oslo, using the software tool developed and documented in Sægrov [52], to analyse historical basement flooding events in Oslo, and make forecasts based on the knowledge gained from such an analysis.

Tool	Sub-system analysed	Aspect of sub-system analysed
MFA	Water pipelines	Pipeline and rehabilitation materials (stocks also included)
	Wastewater pipelines	Pipeline and rehabilitation materials (stocks also included)
	WTPs	Operation-phase chemicals consumption
	WWTPs	Operation-phase chemicals consumption
	Pumping stations	None
EA	Water pipelines	Installation, rehabilitation, operation and maintenance (O &M)
	Wastewater pipelines	Installation, rehabilitation, O & M
	WTPs	Operation-phase energy use
	WWTPs	Operation-phase energy-use
	Pumping stations	Operation-phase energy use
EEA	Water pipelines	Production of pipelines
	Wastewater pipelines	Production of pipelines

Tool	Sub-system analysed	Aspect of sub-system analysed
LCA	Water pipelines	Production, installation, rehabilitation, O & M
	Wastewater pipelines	Production, installation, rehabilitation, O & M
	WTPs	Generation/production and transport/transmission of energy and chemicals; and associated emissions from the use-phase
	WWTPs	Generation/production and transport/transmission of energy and chemicals; and associated emissions from the use-phase
	Pumping stations	Electrical energy consumed during operation phase
Pipeline blockage analysis	Wastewater pipelines	Historical blockage analysis of wastewater pipelines

Tool	**Sub-system analysed**	**Aspect of sub-system analysed**
LCC	Wastewater pipelines	Life-cycle costing optimisation of expenditure and investments in the future, in an asset management perspective
Simple economic/cost analysis	WTPs & WWTPs	Chemicals and energy consumption
	Pumping stations	Energy consumption for pumping

Table 3.1: Overview of the scope and extent of research

3.2 Empirical data acquisition

All the primary empirical data for the Oslo case study have been sourced from Kristiansen [16], Brenden and Berger [17], Reksten [18], Toftdahl [19], BEVAS [20], Aasebø [21], VEAS [22], Selseth [23] and Statistics Norway [24] - also referred to in the first chapter where the Oslo water and wastewater system has been briefly introduced. Interactions with personnel in industry and government, and experts in the academia have proved to be indispensable – as sources of data or as authorities to double-check the accuracy of the secondary data obtained from literature sources.

Raw data (for direct use or as the bases for deriving secondary data) on energy and chemicals consumption, water supplied, wastewater treated and discharged and key by-products generated – the high-frequency inflows into and outflows from WTPs and WWTPs in other words – have been obtained from Toftdahl [19], BEVAS [20], Aasebø [21] and VEAS [22]. For the pumping stations, access to reliable and robust data as regards the influx and outflow of material masses (associated with the pump sets) was difficult. Oslo VAV could provide the author with operational electricity consumption data for a limited number of years – 2005 to 2009 for water pumping and 2007-2009 for sewage pumping. Assuming proportionality to the volumes of water and sewage pumped, respectively, the consumption data for the other years – 2000 to 2004 for water pumping and 2000 to 2006 for sewage pumping have been derived. As gathered from Reksten [18], the energy consumed during maintenance activities at pumping stations has been / is usually negligible in comparison with the operational electricity use.

The pipeline databases obtained from Selseth [23] list the length, material, diameter and year of installation of individual pipe-lengths in operation. There are three diameter classes, respectively, for water and wastewater pipelines. For the former, small (diameter of 199 mm and less), medium (between 200 and 399 mm) and large (over 400 mm); and for the latter, small (249 mm and less), medium (250 to 499 mm) and large (500 mm and greater) are considered by the utility. Further, the said database categorises wastewater pipelines as stormwater, sewage and combined flow pipelines. Records incomplete with regard to diameter and/or material and/or year of installation have been kept out of the analysis. However, they form only a minor part of the pipeline stock. The few stone and brick pipes have been excluded owing to difficulties in obtaining the thicknesses. The other components of the pipeline networks – manhole covers etc. – have not been considered. The masses of the inorganic backfilling material have been calculated using estimates provided by Sægrov [111], which have been outlined in *Appendix 1.*

Embodied energy values for pipeline materials have been sourced from Ambrose [103] and Ambrose et al [62]. The average composition of bitumen used to coat pipelines has been obtained from Andersen [104], in order to split it up into the constituent material inflows. Interactions with Ai [105] and Duke [106] have provided some insights into what utilities think about developing and using sustainable development indicators / indices as metrics to monitor and improve performance. Thicknesses of plastic pipelines – polyvinyl chloride, polyethylene and polypropylene – have been sourced from the industry (Gjersø [107]). Pipeline databases and data on the annual expenditure on the pipeline networks of the cities of Tromsø and Trondheim (needed when the relation between population density and pipeline material mass per capita was investigated, as an incidental offshoot) were received from Helø [108], Jakobsen [109] and Relling and Thue [110]. Courtesy Sægrov [111], data regarding, *inter alia,* the thicknesses of pipelines, and coatings (both protective and rehabilitative) applied to them, have been gathered. Personal communication with Westheim [112] has made available data about the mass, material of construction and lifetime of manhole covers.

Currency exchange rates to convert Norwegian Krone to Euro (Norges Bank [113]), data on power generation in, power imports into and power exports from Norway (as and when required during energy analyses) (Norwegian Ministry of Petroleum and Energy [114]), useful information about polyaluminium chloride or PAX (used as a coagulant in water treatment) (Wikipedia [115]), and price data for energy elements (Statistics Norway [116]) have been useful at different stages of the analyses. In order to modify the Ecoinvent LCA datasets (Swiss Centre for Life-cycle Inventories [39]) for ferrous pipeline fabrication materials, data from Eurofer [117] have been used. The LCA datasets in the Ecoinvent database are usually composed of average values (for material inputs and life-cycle emissions) on national, regional or continental levels. If specific data for the cases being investigated are known, new datasets can be created by replacing the average values in the original datasets with the specific ones.

3.3 Analytical methods – industrial ecology tools

It is worth quoting from Harremoes [28] before moving over to describe the application of the industrial ecology tools in greater detail - 'Life-cycle analysis and mass balances (interpreted as mass flow analyses) will have a much greater effect on our daily life than we can imagine.'

3.3.1 Material stock and flow analysis and energy analysis

As has been said in Brunner and Rechberger [29], 'MFA is of prime importance for analysis and planning. It is the basis of modelling resource consumption as well as changes in stocks, and therefore it is important in forecasting the scarcity of resources. It is helpful in identifying the accumulation and depletion of materials in natural and anthropogenic environments.' The analysis of energy and material flows has come to be recognised as an important (and often a first) step in reducing the human impacts on the environment (Binder [30]). MFA can be labelled as the precursor to material flow management – one of the many functions bracketed under the umbrella-term 'Asset Management'. It is also a vital first step in quantifying a system's environmental impacts, which necessitates the knowledge of the type and magnitude of material and energy inflows and the associated emission / waste outflows. The material and energy flows for the water and wastewater network as a whole, can be split up into flows for the pipeline networks, pumping stations, and the water and wastewater treatment plants (WTPs and WWTPs). An integration may be done, if need be, after the individual analyses are completed, to understand the diversities with respect to the types, quantities and frequencies of material flows.

Studying the material flows is concomitant with understanding the build-up of the stocks. In the case of the pipeline networks for instance, understanding the stock composition entails a thorough foreknowledge of the characteristics of all the pipe elements in the network. Pipelines can be categorised primarily on the basis of length, year of installation, diameter, function and material of construction. The year of installation and material of construction together will also give an insight into the technology adopted to fabricate the pipeline, and the expected useful lifetime of the same. This knowledge is indispensable when rehabilitation strategies (and also maintenance schedules) are drawn up (Ugarelli et al [25]) or when an LCC analysis is done to perform a pipeline blockage analysis (*Section 3.4).* A stock with a preponderance of old pipelines nearing the end of their respective useful lives clearly indicates the need for greater rehabilitation in the near future. Subsequently, the rehabilitation material inflows can also be forecast. Knowing the composition of the pipeline stock at a given point in time also enables one to chart not just the historical flows which have built up the stock, but also the environmental impacts associated with these flows. While the stock of water and wastewater pipes in the built environment of the city of Oslo is analysed as part of this research, Brattebø et al [33] have applied the method to bridges and buildings in Norway, inspired by thoughtful proposals on how to better structure fundamental research on systems in the built environment (Kohler and Hassler [118] and Kohler and Yang [119]).

The stock of water and wastewater pipelines in Oslo, is composed of significant amounts of materials – pipeline materials like concrete, ductile iron, grey cast iron, mild steel, polyvinylchloride, polyethylene, stone and brick, copper, asbestos cement, etc., coating materials like bitumen, cement mortar, zinc and galvalume (an alloy of zinc and aluminium), and rehabilitation materials like epoxy resin and polyurethane. Backfill (or bedding) material – sand and crushed gravel – has been / is employed during installation. Diesel fuel has been / is consumed during the installation, operation and maintenance (O&M), rehabilitation and retirement processes. Electricity consumed in the administrative offices has not been taken into consideration in this analysis. Besides, allocating this consumption among the different components of the system is difficult. Rehabilitation entails inflows of epoxy resin and polyurethane (PU). The Cast-in-Place-Pipe (CIPP) approach has been (and still is) the most prevalent among rehabilitation methods (Ugarelli et al [25]). PU is a more recent introduction as compared to epoxy resin (Sægrov [111]). In this analysis, for the sake of simplification, it has been assumed that all rehabilitation was carried out using epoxy resin.

The combustion of diesel results in gaseous emissions. Pipelines, when 'retired' at the end of their lives, are disconnected from the network and left buried beneath the ground, potentially also contributing to what has been referred to earlier (Mueller et al [61]) as the urban 'mines of the future'. As far as the coatings are concerned, the 'outflows' are essentially dissipative losses to the water being transported, and also to the lithosphere / pedosphere.

Using values for material thicknesses obtained from Gjersø [107] and Sægrov [111] (tabulated in *Appendix 1*), standard specific gravities for the materials (tabulated in *Appendix 2*), and the diameters and lengths known from the pipeline databases, the pipeline material mass inflows have been calculated. Here, it has to be pointed out that the diameters recorded for the plastic pipes are the outer diameters, while those recorded for pipes of all other materials of construction are the inner diameters. The consumption of diesel has been calculated on the basis of estimates of per-metre-pipeline consumption values for installation and rehabilitation (*Appendix 3*) and absolute values for consumption during the O&M phase obtained from Kristiansen [16]. Owing to non-availability of accurate data about the water and wastewater pipelines which have been disconnected and 'retired' from the network and replaced by new pipe-lengths, these 'outflows' have not been determined. For the same reason, the energy expenditure for the retirement process has not been estimated. However, these 'outflows' and thereby, the mass inflows by way of pipe-replacement, have been small enough for one to assume that almost all the pipeline material mass inflows have contributed to the expansion of the pipeline networks. As far as the data for rehabilitated water pipelines are concerned, the database has some datasets for which the year of rehabilitation has not been recorded. These have been kept out of the analysis.

Owing to data insufficiency for the period before 1991 - as regards material and energy flows in the rehabilitation and O&M phases - the comprehensive material and energy flow analyses have been carried out only for the years 1991 to 2006. A forecast of possible material flows into the wastewater pipeline network has been done for the two-decade period 2008 to 2027 as a precursor to an LCC performed in Venkatesh et al [120]. *Figure 3.1* represents the material flows into and out of, the energy flows into, and the emissions out of the water and wastewater pipeline network in Oslo.

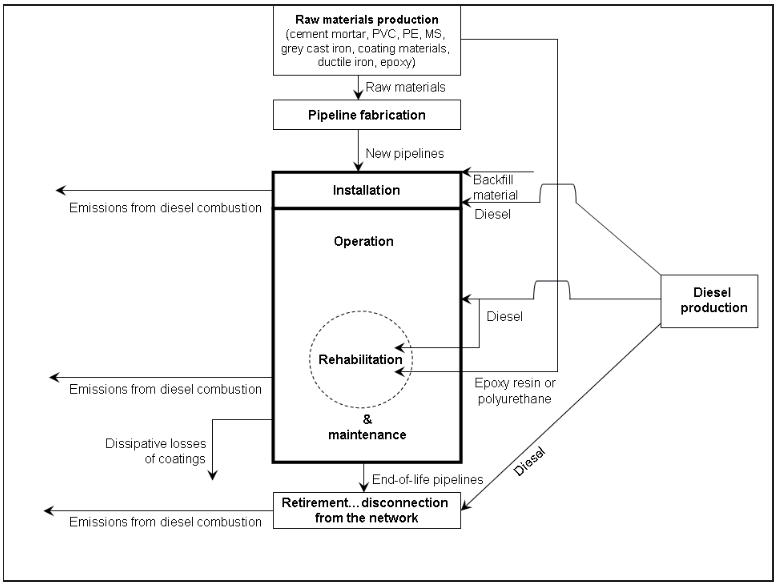

Figure 3.1: Schematic depiction of material and energy flows into and out of Oslo's water and wastewater pipeline network

As far as the material and energy flows in water and wastewater treatment are concerned, one can differentiate between the flows associated with the initial construction and the final demolition phases on the one hand, and those which occur during the intermediate operation-maintenance-refurbishment-overhauling phase on the other. The former are the low-frequency flows, while the latter constitute the high- and medium-frequency ones. The principal inflows in the operation phases of WTPs and WWTPs are the raw water and untreated wastewater respectively, while the principal outflows are the treated water (for supply) and the treated effluent wastewater (for discharge). Chemicals, materials and energy are pressed into service to keep the plants performing their functions of modifying the influent (water or wastewater as the case may be) for the desired purposes.

Obtaining primary data and estimating the inflows of building materials, equipment and machinery spares, electricity and energy carriers during the construction (overhauling, refurbishment, maintenance and expansion) phases was difficult. The same was true for the medium-frequency and low-frequency outflows of building material wastes, equipment retired from service, replaced machine spare parts sent for reuse/recycling, and the emissions associated with the consumption of fuels during the construction, O&M and demolition phases.

However, we believe, but have not been able to document, that these elements of the life-cycle are of relatively less importance.The focus thereby had to be restricted to the flows related to chemicals and energy consumption and the key by-products of treatment processes *(see Appendix 4)*. This is a limitation of this research, which should (and will) be overcome in subsequent follow-up work.

It needs to be mentioned at this juncture that the VEAS WWTP handles wastewater from Oslo as well as from some of the neighbouring municipalities. A proportional allocation of the inflows and outflows has thereby been done, on the basis of the shares of the different municipalities in the total volume of the influent wastewater. The shares were obtained from VEAS [22]. There are some non-consumptive flows in the WWTPs like those of sea-water (used as a heat-exchange fluid) and clean water for filter-backwashing. The values of these flows have been recorded only for VEAS (and were not available for BEVAS at the time of the study). The consumption of air (oxygen) during the treatment processes has not been recorded in the databases. The atmospheric outflows associated with on-site fossil energy consumption have not been explicitly indicated, though these have been automatically accounted for, in the LCA. Inflows, in general, are partitioned at the treatment plants (WTPs and WWTPs) into the solid (sludge and screenings), gaseous (emissions to the atmosphere) and liquid (discharge with the treated water or wastewater) streams. The partitioning cannot be done with a high degree of accuracy for all the inflows into the WTPs and WWTPs. Valid and justified assumptions have been made on this front, to enable the determination of the environmental impacts associated with their operation phases.

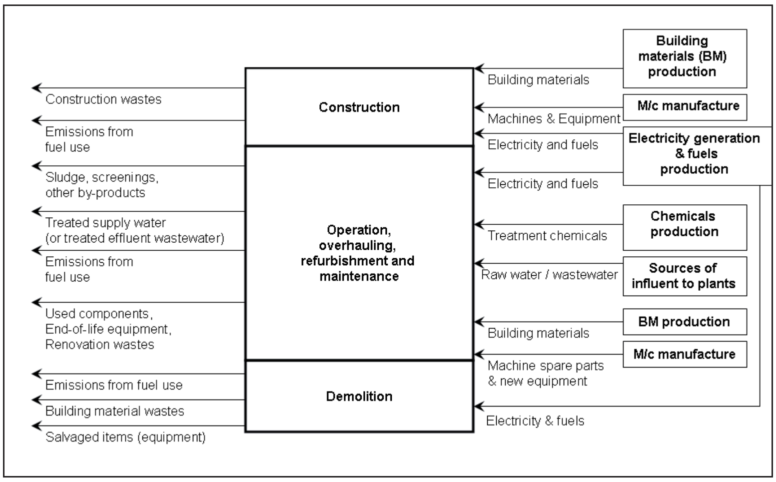

Figure 3.2: Schematic depiction of material flows into and out of, energy flows into, and emissions out of Oslo's water and wastewater treatment plants

Figure 3.2 depicts schematically the life-cycle material flows into and out of, energy flows into, and emissions from WTPs and WWTPs in Oslo. It goes without saying that between WTPs and WWTPs, there are major differences in the chemicals and energy inflows and the associated outflows, both quantitatively and qualitatively.

3.3.2 Environmental life-cycle assessment

LCA, as Rebitzer et al [42] has pointed out, supports the identification of opportunities for pollution prevention and reducing resource consumption through systematic analysis and avoiding dogmatic objectives which can be, while intuitive, incorrect even in their general tangent. If an LCA has to be carried out to determine the changes in annual environmental impacts over time, one would begin with an MFA and an EA, in order to determine the flows of the agents responsible for the environmental impacts. The MFAs and EAs carried out (referred to in the previous section), have served as the bases for subsequent LCAs – of the water and wastewater pipeline networks, the energy and chemicals consumption, and associated emissions from WTPs and WWTPs, and energy consumption for pumping water and sewage.

The construction, refurbishment-maintenance-expansion and demolition phases of the WTPs, WWTPs and pumping stations, and the retirement phase of the water and wastewater pipelines, have not been taken into consideration. As pointed out in Sahely and Kennedy [98], the operational environmental impacts are much more important for the key environmental indicators than capital infrastructure over the life-cycle of urban water systems, though Vince et al [67] have stated that the contribution of the construction phase to mineral resource depletion may not be insignificant. Recourse has been taken to PRé Consultants [37], CML-University of Leiden [38], Swiss Centre for Life-cycle Inventories [39], Huijbregts et al [40] and Building and Fire Research Laboratory, National Institute of Standards and Testing, USA [41] (all called out to, for the first time in the second chapter) for all the LCA studies forming a part of this research.

The methodology adopted for the LCA studies, in keeping with the ISO 14040 set of standards (Rebitzer et al [42]) is as under -

- Goal and scope definition
- Life-cycle inventory analysis
- Life-cycle environmental impact assessment
- Interpretation

It has been described in detail in Venkatesh et al [121], and hence not repeated in this chapter. Only six midpoint impact indicators have been taken into consideration in the analyses – abiotic depletion, acidification, eutrophication, global warming, ozone depletion and photochemical oxidant / ozone creation. The measured impact potentials – the midpoint indicators - for these six impacts are reasonably close to the corresponding real potentials. The differences can be attributed to several influences which have subtractive or additive effects on the measured potentials. On the other hand, owing to the uncertainties inherent in the fate-exposure-effect modelling for emissions contributing to marine/freshwater/human toxicities, the deviations of the measured toxicity potentials expressed as midpoint impact indicators, from the real potentials are much greater. It is for this reason that the toxicity categories have not been included in the analysis.

The CML 2002 Impact Assessment Method (CML, University of Leiden [38]) is adopted. The normalisation factors (West European, 1995; Huijbregts et al [40]) and the weighting factors (prescribed by the United States Environmental Protection Agency, BFRL [41]) used are tabulated in *Table 3.2.* The use of the USEPA-recommended weighting factors referred to, may be debatable when used in a European context. It is however reasonable to assume that, to a great extent, the relative importance of the six impact categories with respect to each other, is the same for all western-world settings. It is a known and accepted fact that weighting factors are closely related to national or regional policies, governmental regulations and political decision-making. If Norwegian, Scandinavian or European weighting factors had been available, any of these would have been more apt than the USEPA-prescribed factors. Further, communicating results in an LCA becomes easier when a set of impacts is whittled down to a single number – after normalisation, weighting and aggregation.

Impact category	Normalisation factor	Weighting factor
ADP	1.48E+10	5
AP	2.43E+10	5
EP	1.25E+10	5
GWP(100)	4.81E+12	16
ODP	8.33E+07	5
POCP	8.22E+09	8

Table 3.2: Normalisation factors and weighting factors adopted for the LCA analyses

On the other hand, one would expect that qualified strategic decisions by well-informed stakeholders, would necessitate a more in-depth evaluation than using aggregated single-number LCA indicators. In such cases, the stakeholders (decision-makers in other words) would possibly have their own preferences for selected environmental impact categories. As far as urban water and wastewater systems are concerned, the main concern may be water pollution issues (eutrophication potential, in that case becoming a key impact), or energy consumption and GHG emissions (bringing abiotic depletion and global warming potentials to the forefront) or it may be the influence of water quality by acid rain (acidification potential assuming paramount importance). This research has not made an attempt to find out what the actual preferences of Oslo VAV and the numerous water system stakeholders in Oslo are. However, it must be mentioned at this juncture that when LCA is incorporated into strategic policy-making, this is a pre-requisite.

For water and wastewater treatment, the impact scores have been expressed in terms of per-unit-volume of water supplied and wastewater treated. On the basis of environmental impacts (impact scores in other words), water pipeline networks have been compared with wastewater pipeline networks in Venkatesh and Brattebø [122] for the period 1991-2006, a *vis-à-vis* between energy consumption in water treatment and wastewater treatment has been done, for the period 2000-2007, and between energy consumption and chemicals consumption for water treatment and wastewater treatment separately in Venkatesh and Brattebø [123] and [124]. In Venkatesh et al [121], future environmental impacts attributable to the wastewater pipeline network have been forecast, based on results obtained in Venkatesh et al [120]. The environmental impacts associated with energy consumption for water and sewage pumping, have also been determined, and the different elements of the system have been compared with each other.

3.3.3 Life-cycle costing

The LCC methodology, as referred to earlier, has been explained in Fuller [44]. In Rebitzer, Hunkeler and Jolliet [89], the LCA-based LCC tool is posited as a powerful one in expanding the economic view while studying wastewater treatment. Rebitzer et al [42] state that LCC can be effectively combined with LCA in order to represent two of the three pillars of sustainable development – the environment and the economy.

Referred to variously as whole-life costing analysis, womb-to-tomb costing analysis, or cradle-to-grave costing analysis, LCC can very simply be defined as the total cost of ownership over the lifetime of an asset. Expenditure and income streams of the future are expressed in terms of their present value, by considering discount rates (a sensitivity analysis is also possible here), and subsequently the net present value (NPV; the difference between the total present value of all income streams and the total present value of all expenditure streams) is calculated.

As shown in *Figure 3.3,* the income and expenditure streams in real 'year-0' currency units (adjusted for an expected interest (discount) rate, with respect to year '0') are identified for every year of the life-cycle. The discount rate, as indicated, is r %, and year '0' in which the capital investments to construct the system are made, is considered as the year to which all the incomes and expenditures are to be discounted back, to determine the NPV. If C_n stands for the costs or expenditures in year 'n' and B_n stands for the income or benefits in year 'n', then the NPV is calculated as below (assuming in this instance, a nine-year lifetime, and no operation in the year labelled '0', during which only capital investments occur)

$$NPV = -C_o + \sum_{n=1}^{n=9} \frac{(B_n - C_n)}{(1+0.01*r)^n} \qquad \text{(Eq. 3.1)}$$

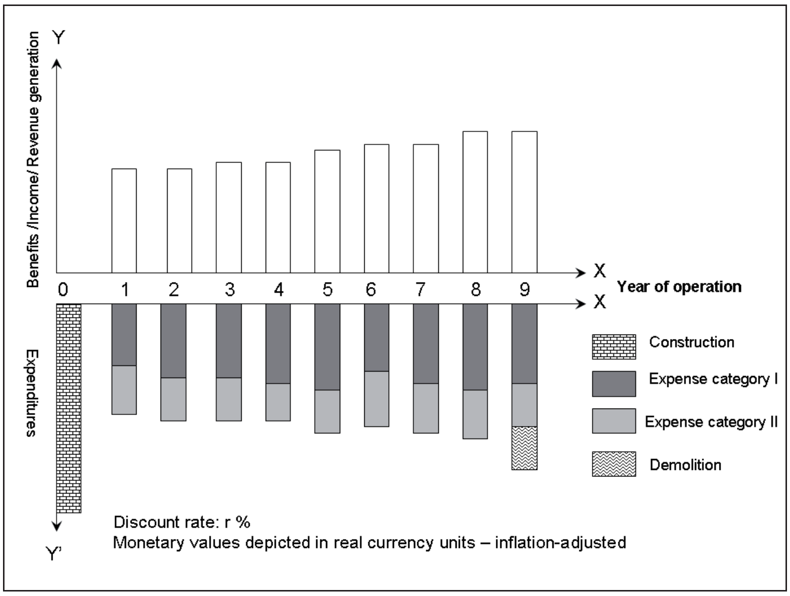

Figure 3.3: Life-cycle costing analysis explained graphically

A rational decision-maker looking for maximising gain would opt for the alternative with the highest NPV. The purpose of performing an LCC would ideally be to determine the most profitable investment- expenditure alternative available. While an LCA would enable one to select the most environment-friendly option, an LCC would identify the most economically-attractive option (the one yielding the highest monetary benefits). Treating these two approaches in isolation may result in two conflicting end-results. Assigning costs to environmental externalities would integrate the two approaches and obviate a sacrifice of the environmental benefits at the altar of the economic ones, particularly if future environmental (indirect) costs are to be discounted using the same discount rate as the direct project costs. However, these issues are more in the realm of political decision-making, and this research, even while it acknowledges and recommends an integration of the two approaches, has not explicitly attempted to combine the two.

While the LCA tool has not been employed for comparing alternate approaches to identify the best one, but just to estimate the environmental impacts over time (a descriptive attributional LCA in other words), the LCC tool has been used to prove in Ugarelli et al [25] and Venkatesh et al [120] that the recommended rehabilitation approach (based on physical lifetimes of pipelines) for wastewater pipelines in Oslo is better than the in-vogue one, which is based on the economic lifetime. Subsequently, it has also been shown in Venkatesh et al [121] that this rehabilitation approach is preferable over the existing one, not just from an economic, but also from an environmental point of view. The economic data on the investments and O&M expenses, which constitute the basis of Ugarelli et al [25] and Venkatesh et al [120], were obtained through interaction with Kristiansen [16]. It must be mentioned at this juncture that water and wastewater pipelines typically have long lifetimes (longer than the nine years depicted in *Figure 3.3*), and the end-of-life demolition costs are almost negligible. The expected costs in later years, when discounted back to the year of installation (year '0' in other words), would not contribute much to the NPV.

A study on the lines of Ugarelli et al [25] has not been performed for the water pipeline network, owing to lack of timely access to economic data. There is great scope for applying LCC to study the impacts of possible process modifications on the economics of water and wastewater treatment.

3.3.4 Embodied energy analysis

EEA is not very different from the life-cycle energy consumption (or EA in other words). However, while in the case of an EA (which more often than not, is one of the precursors to a life-cycle environmental analysis), the total energy consumption is broken down into the different energy carriers (and the associated upstream processes); in an EEA, no distinction is made for instance, between a kilowatt-hour of electrical energy and a kilowatt-hour of heat energy, or for that matter, a joule of renewable energy and a joule of non-renewable energy.

An EEA has been attempted for the water and wastewater pipeline networks. This analysis has been performed primarily to compare the per-capita embodied energy values for water and wastewater pipeline networks in Oslo, with those of two other Norwegian cities Trondheim and Tromsø, at a given instant in time. Material production, pipe fabrication and diesel consumption during pipe installation have been taken into consideration. The energy consumption associated with the processing and transport of the backfill (bedding) material has not been considered. Likewise, the coating materials have also been excluded from the analysis. The subsequent phases in the life-cycle of the pipelines – rehabilitation, O&M and retirement - are beyond the scope of embodied energy as defined in this study, and thus, epoxy resin and polyurethane have been ruled out.

3.4 Pipeline blockage analysis

During the maintenance phase of wastewater pipelines, blockages are often to be contended with. There is a strong correlation between blockages of pipelines and basement flooding events in Oslo. The utility has to monetarily compensate the households afflicted by such flooding events, in addition to expending on reactive maintenance. Preventive/proactive maintenance or more preferably, timely rehabilitation of pipelines will enable the utility to curtail the incidences of blockages and thereby basement flooding. A historical analysis of blockage-induced flooding events will enable the utility to gauge the proneness of pipelines to such failures, on the basis of their characteristics – diameter, material, age, location and slope of installation, function performed in the network, hydrological details, etc. This will facilitate perfectly-optimised maintenance and rehabilitation strategies. The Material Stock Analysis described in an earlier section, is the starting point of the pipeline blockage analysis which has been carried out using the CARE-S Blockage Tool (Sægrov [53]) to study the blockage events between the years 1991 and 2006 and the distribution thereof among the different categories of pipelines in the wastewater pipeline network of Oslo (Ugarelli et al [125]). This tool calculates a blockage probability factor for each pipe by using a factorial-based model where the influences of different explanatory factors such as pipe material, wall thickness, length, installation year etc. are accounted for.

Before running the tool, pipes were grouped, and thus each pipe belongs to a specific class within each group. Groupings were done on bases of material, age/installation year, diameter, sewer type, pipe slope, surrounding soil, backfill material, CCTV inspection results or condition class, shear stress/velocity, flow (from hydraulic model), and pipe position in the network (from GIS analysis). The blockage factor for the class equals the blockage rate for the class (blockages per kilometre of pipe per year) divided by the blockage rate for the whole network. In this particular case, the time period considered was 16 years (1991-2006). Hence, the blockages per kilometre of pipeline of a particular class, had to be divided by 16, in order to obtain the blockage rate in the desired unit.

4. Results

The results can be grouped under three sections

- Water distribution and wastewater transport
- Water and wastewater treatment
- Overview of the entire system

As mentioned earlier, all the tools have not been applied to study all the components of the system. Some of the results bear reference to published papers (or papers which have been accepted for publication or are under review). Others originate from unpublished studies, not under review with any scientific journal. The papers appended at the end of this thesis, have been listed at the beginning (*page xiii*), as an easy reference for readers.

4.1. Water distribution and wastewater transport

Transport and distribution essentially include the pipelines and the pumping stations (refer *Figure 1.2*), in addition to other components (Ugarelli et al [25]) which have not been included in this analysis.

4.1.1. Pipeline networks

Both water and wastewater pipeline networks have been dealt with, in this sub-section. The development in the stocks of pipelines has been studied with respect to size, type (function performed) and material of construction, for the years 1900 to 2006. Such an analysis which enables one to understand the phenomena of ageing and saturation in pipeline networks is extremely useful in asset management. The focus has then been narrowed down to the years 1991-2006 for a more detailed analysis. For the said sixteen-year period, energy analysis and embodied energy analysis (EA and EEA), life-cycle costing (LCC) and pipeline-blockage analysis for the wastewater pipeline network and environmental life-cycle assessment (LCA) for both the pipeline networks (based on the results of the material flows, stocks and energy analyses) have been performed. Selected results from each of these studies are presented below.

Lengths: At the end of year-2006, of the functional water pipelines of known diameters, the large (diameter >=400 mm), medium (200 mm <= diameter <= 399 mm) and small (diameter <=199 mm) accounted for about 183 kilometres (km), 524 km and 758 km respectively. That amounted to a total of 1465 km (equivalent to 2.67 metres per capita). In the case of wastewater pipelines, the corresponding numbers were 348 km (diameter <=249 mm), 580 km (250 mm >= diameter <= 499 mm) and 1014 km (diameter >=500 mm), totalling up to 1942 km (equivalent to 3.54 metres per capita). If categorised into sewage carriers, stormwater pipelines and combined flow pipelines, the respective lengths were 656 km, 766 km and 800 km respectively – a total of 2222 km. The difference in the two sums (2222 km – 1942 km) can be explained by the existence of incomplete datasets in the database – while the function of each pipeline is known, there are some for which either the diameter or the length is not registered. The evolution of the functional stock of water pipelines is depicted in *Figure 4.1.* That of the wastewater pipelines has been described in detail in Ugarelli et al [25] and also depicted in *Figure 4.2* and *Figure 4.3.*

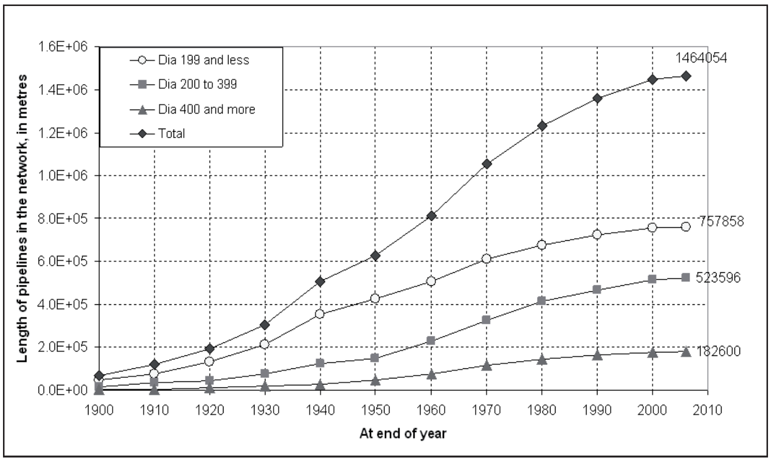

Figure 4.1: Growth in stocks of water pipelines in Oslo (length)

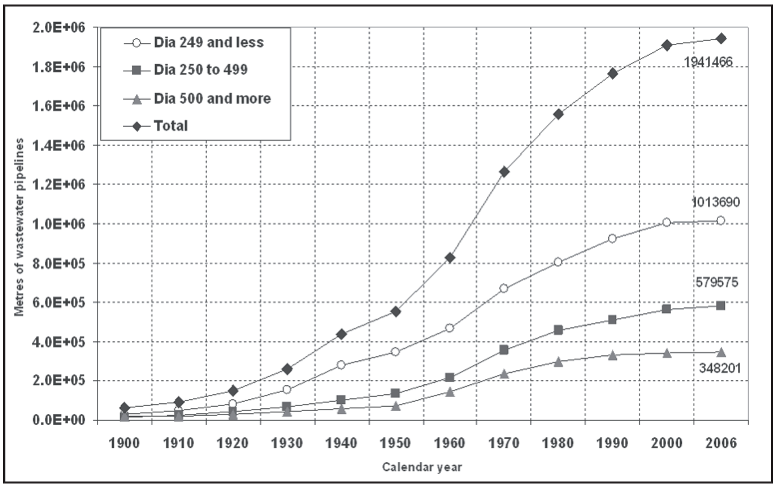

Figure 4.2: Growth in stocks of wastewater pipelines in Oslo (length)

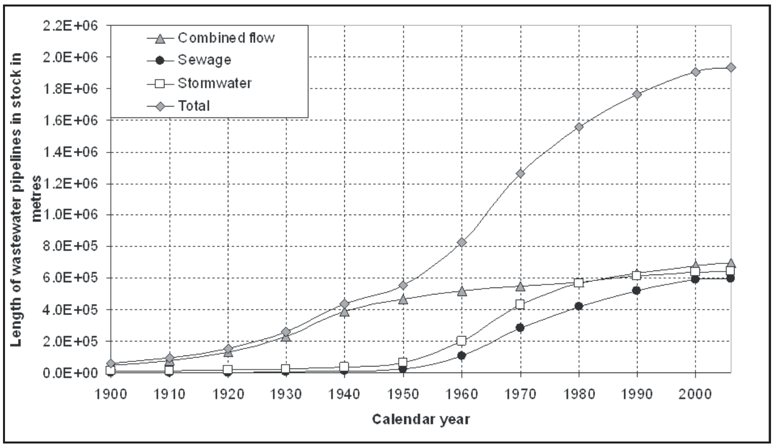

Figure 4.3: Growth in stocks of wastewater pipelines in Oslo, in terms of (type) function in the network

From *Figure 4.3*, it is seen that the separation of wastewater flows into stormwater and sewage, started in the 1950s; and since then, the share of stormwater and sewage pipelines in the network has increased at the expense of combined flow pipelines. The addition to the stocks was the most rapid in the twenty-year period 1960-1980, for both the water and wastewater pipeline networks. The saturation phenomenon – in absolute terms (metres) and specific terms (metres per capita of population serviced) - is observed in both the networks after year-1990. As a pipeline network advances towards saturation, its capacity utilization also increases. While this augurs well on the one hand, it also entails a greater (and more frequent) loading of the pipelines, leading to the possibility of earlier-then-expected failures. The per-capita value for wastewater pipelines peaked to a little over 3.8 metres in the early 1990s, before the decline started. The corresponding peak for water pipelines occurred at about the same time, and was a little over 3 metres per capita. It is interesting to note that the per-capita value for water pipelines was greater than that for the wastewater pipelines, till the early 1960s. Thereafter, with wastewater collection and treatment assuming increasing importance in Oslo (and also the world-over at the same time), separation was deemed to be essential and this resulted in a spurt in the addition of pipelines into the wastewater pipeline network, and as a consequence, a rise in the per-capita value for the same over that of the water pipeline network. It is likely that cities with advanced and well-developed wastewater collection and treatment systems will exhibit a similar relationship between the water and wastewater pipeline networks. However, the opposite would be true for cities in the developing world where wastewater separation and treatment are still not as critical as providing water to ever-increasing populations.

At the end of year-2006, the average age of the water pipeline network was 51 years, while that of the wastewater pipeline network was 37 years. The older a network gets, investments in rehabilitation are called for, in order to rejuvenate the network to ensure an upkeep of the required level of service, decrease the leakage rates and reduce the incidences of pipe failures.

During the period 1991-2006, 91 km of wastewater pipelines were rehabilitated, and 172 km of new pipelines were added. Of the total rehabilitated, in terms of length, small-diameter pipelines accounted for 63%, the medium-diameter ones for 30% and large-size ones for the remainder. Of those added to the network, the shares of sewage, stormwater and combined flow were 45%, 14% and 41% respectively. Small-diameter pipelines dominated the additions with 52% of the total kilometres, followed by medium-diameter pipelines (41%) and the large-size ones (7%). Of the 106 km of water pipelines added to the network during the period 1991-2006, and the 47 km of rehabilitated water pipelines which have been considered for this analysis for the said time-period, the small, medium and large-size categories accounted for 32%, 53% and 15% respectively in the case of additions, and 14%, 55% and 31% respectively in the case of rehabilitations.

Mass inflows: *Figure 4.4* and *Figure 4.5* translate the increase in the lengths of pipelines in the water and wastewater pipeline networks into the corresponding increase in the masses of pipeline materials of construction. The mass units are not the same for all the materials owing to vast differences in their shares in the networks.

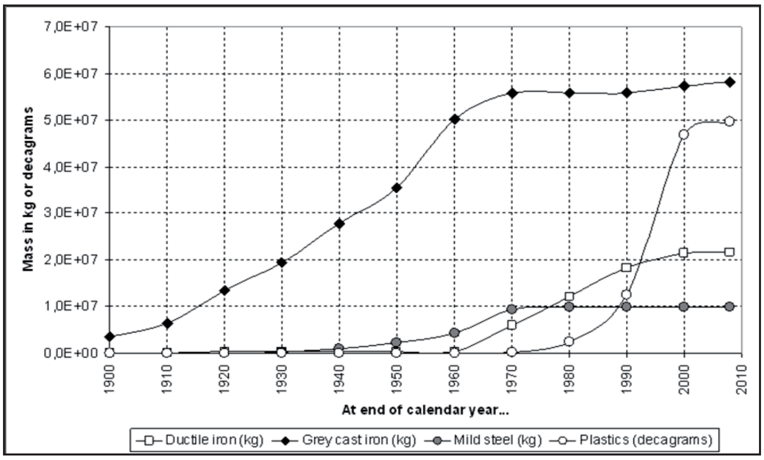

Figure 4.4: Increase in the mass of materials in water pipeline stocks from 1900 to 2006 (Units in kilograms and decagrams)

Grey cast iron has dominated the water pipeline network all through, followed way behind by ductile iron. Mild steel additions ceased in year-1970 – the same time when the incursion of plastic pipes (PVC and PE) started. While plastics accounted for a very small share of the network in terms of mass – a little over 0.5%, grey cast iron made up a little over two-third of the total, ductile iron about 22% and mild steel approximately 10%. The counterpart of grey cast iron in the wastewater pipeline network is concrete. In year-2006, there were over 300,000 tonnes of concrete in the network (Venkatesh et al [121]).

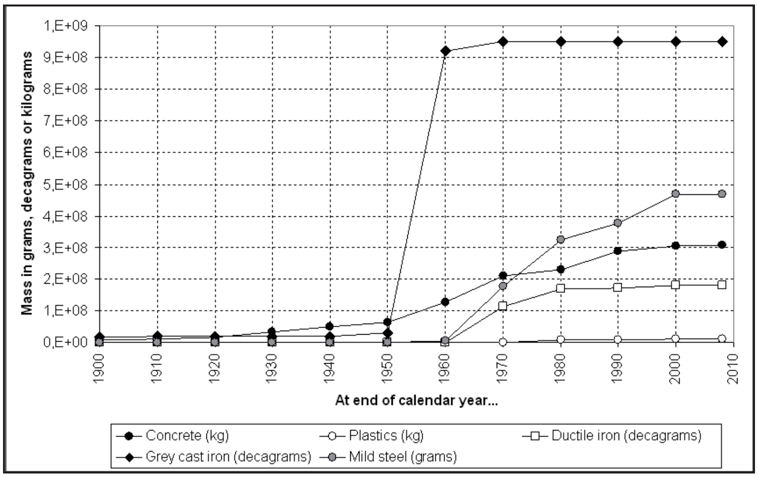

Figure 4.5: Increase in the mass of materials in wastewater pipeline stocks from 1900 to 2006 (Units in grams, decagrams and kilograms)

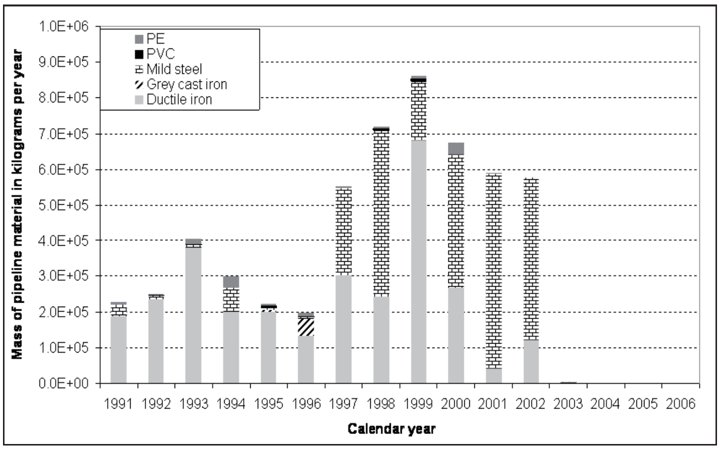

Figure 4.6: Mass inflows of pipeline materials of construction into Oslo's water pipeline network between 1991 and 2006

The annual material flows – pipeline materials of construction and epoxy resin used for rehabilitation – for the wastewater pipeline network – for the period

1991-2006 have been discussed in detail in Venkatesh et al [121]. *Figure 4.6* and *Figure 4.7* depict the pipeline material flows in kilograms per year for the said period, for the water and wastewater pipeline networks respectively. *Table 4.1* shows the mass inflows of epoxy resin into the two networks by way of rehabilitation, while *Table 4.2* lists the inflows of backfill or bedding material (crushed gravel) into the pipeline networks during the installation phase.

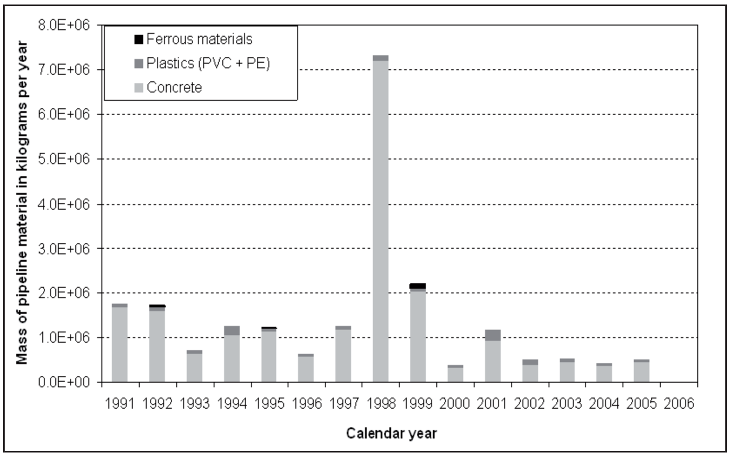

Figure 4.7: Mass inflows of pipeline materials of construction into Oslo's wastewater pipeline network between 1991 and 2006

A total mass of about 5,600 tonnes (of ductile iron, grey cast iron, mild steel, PVC and PE taken together) was added to the water pipeline network in the said period, vis-à-vis 21,700 tonnes (of concrete, grey cast iron, ductile iron, mild steel, PVC and PE taken together) into the wastewater pipeline network. Thus, of the total incursion of pipeline material masses into the pipeline networks in the years 1991-2006, the water pipeline network accounted for 20.4%. Ductile iron commanded the largest share (54% of the mass inflow) in the case of the water pipeline network, and mild steel followed close on its heels with 43%. For the wastewater pipeline network, it was concrete which dominated, with 92% of the total. Taken together, concrete accounted for 73% of the total mass inflows, ferrous materials 21% and plastics 6%. The peak inflow into the water pipeline network during the said time period – about 861.9 tonnes - occurred in year-1999 and accounted for 15% of the total. For the wastewater pipeline network, the corresponding year was 1998, when 7,300 tonnes (33.6% of the total mass inflow into the network) were added. When both networks are taken together, 30% of the total mass inflows occurred in year-1998.

Vis-à-vis the pipeline material masses, the epoxy resin inflows were quite small. About 220.6 tonnes of epoxy resin were introduced into the water pipeline network in the said period, compared to 573.2 tonnes (2.6 times more) into the wastewater pipeline network. These were 29 and 37 times less than the mass inflows of pipeline materials into the respective networks. A look at the last column in *Table 4.1* reveals that till the turn of the century, rehabilitation of wastewater pipelines dominated the flow of epoxy resins into the networks. From year-2001, it was the other way round. The peak epoxy inflow in the case of the water pipeline network happened in year-2002 (38.9 tonnes or 17% of the total), while for the wastewater pipeline network, the peak occurred in year-1991, with 128.6 tonnes (or 22% of the total).

Calendar year	Into the water pipeline network (tonnes)	Into the wastewater pipeline network (tonnes)	Total (tonnes)	Ratio (Wastewater / Water)
1991	1.74	128.60	130.34	73.91
1992	0	113.37	113.37	------
1993	0.48	39.71	40.18	83.59
1994	5.27	49.44	54.72	9.38
1995	0.54	29.68	30.21	55.47
1996	0	30.91	30.91	-------
1997	13.20	31.38	44.58	2.38
1998	29.85	32.49	62.33	1.09
1999	11.34	25.62	36.96	2.26
2000	23.41	23.54	46.95	1.01
2001	30.22	22.67	52.89	0.75
2002	38.92	15.77	54.69	0.41
2003	22.78	10.44	33.22	0.46
2004	17.26	7.37	24.63	0.43
2005	24.39	12.22	36.61	0.50
2006	1.18	0	1.18	0.00

Table 4.1 : Epoxy resin annual mass inflows into the water and wastewater pipeline networks in Oslo during the period 1991-2006.

Crushed gravel which is used as a backfill material during the installation of pipelines is a prime input into the networks, in terms of mass. About 2.6 million tonnes of backfill material were added to both the networks taken together, during the said time period (The calculations are done based on data provided by Sægrov [111]; refer *Appendix 1(e)*). The wastewater pipeline network accounted for a much larger share than the water pipeline network, owing obviously to the fact that more kilometres of wastewater pipelines were installed as compared to water pipelines. The peak inflow occurred in year-1991 – 316,000 tonnes – and the average annual inflow during the period was 165,000 tonnes. Vis-à-vis the inflows of the pipeline materials of construction, epoxy resin (used for rehabilitation) and backfill crushed gravel, the additions of bitumen, cement mortar and zinc as coating materials, were very small. *Table 4.2* also lists the mass of bedding material used per kilogram of pipeline material introduced into the network in the said period. On an average – with 2.6 million tonnes for

27,300 tonnes of pipeline materials of construction – the specific value comes to 95.2 kilograms per kilogram.

Calendar year	Total backfill mass inflow ('000 tonnes)	Ratio of backfill material mass to pipeline material mass (kg/kg)	Ratio of backfill mass inflows into the water pipeline network to those into the wastewater pipeline network
1991	316	159	0.015
1992	286	146	0.000
1993	164	145	0.011
1994	194	124	0.076
1995	114	78	0.018
1996	102	123	0.000
1997	174	96	0.266
1998	206	25	0.778
1999	174	57	0.304
2000	167	155	0.575
2001	163	92	0.896
2002	202	185	0.990
2003	125	229	0.619
2004	92	213	0.894
2005	135	256	0.749
2006	39	3549	0.083

Table 4.2: Mass inflows of backfill material during installation of pipelines into the networks in Oslo (1991-2006). (Refer *Appendix I (e)* for the mass calculation details)

While that was a summary of the mass inflows into the pipeline networks between the years 1991 and 2006, it must be reiterated that 'outflows' *per se* are difficult to estimate. The term 'outflows' is a misnomer, when one takes into account the fact that pipelines, after retirement, become inactive and are usually left *sub-terra.* This may or may not be considered to be a flow out of the system. 'Out of the system' in this case would be tantamount to an extraction above the ground for possible recycling. Besides, while bearing in mind that retirement is followed by a replacement of the corresponding pipe-length (in effect, a rehabilitation process which adds pipeline materials to the network), the annual rehabilitation rates for both water and wastewater pipeline networks during the years 1991-2006 were not significant (less than 1% on average; calculated from the database supplied by Selseth [23]), and replacements only formed a small subset of rehabilitation (as gathered from Sægrov [111]), the material masses which were rendered inactive annually, are negligible. It can also be inferred thus that almost all the pipeline material mass inflows into the network were consequences of network expansions. There are always dissipative losses of the coating materials and also of the crushed gravel, but these are very difficult to estimate. *Table 4.3* charts the per-capita mass values for pipeline and coating materials in the stock of operational water and wastewater pipelines, as at the end of year-2006, when the population of Oslo was 548,617. Epoxy resins have not been included owing to inability to acquire data about rehabilitation for the pre-1991 period of the lifetime of the pipeline networks. Material additions to the pipeline networks, in the form of additional pipelines, have been negligibly small in the years 2006 - 2010 (as gathered from Kristiansen [16]). The population has increased slightly, and hence, the numbers for year-2010 may be slightly less than the ones shown in *Table 4.3* for year-2006.

Material	**Water pipelines**	**Wastewater pipelines**	**Total**
Pipeline materials of construction (kg)	**163.9**	**600**	**763.9**
Concrete (and asbestos cement)(kg)	0	559	559.0
Ductile iron[1](kg)	39	3.2	42.2
Grey cast iron[2](kg)	106	17	123.0
Mild steel[3](kg)	18	0.85	18.9
PVC, PE & PPP(kg)	0.9	20	20.9
Coating materials (kg)	**15.8**	**1.12**	**16.92**
Bitumen(kg)	9.8	0.93	10.7
Cement mortar(kg)	6	0.19	6.2
Zinc(decagrams)	6.7	0.22	6.9
Elemental analysis (pipeline materials)			
Iron (in ferrous alloys and in reinforcements)(kg)	156	20.5	**176.5**
Carbon(kg)	3.5	0.46	**3.9**
Silicon(kg)	3.8	0.43	**4.2**
Phosphorus (decigrams)	78	6.5	**84.5**
Sulphur (decigrams)	90	4.3	**94.3**
Manganese (grams)	320	19.9	**339.9**
Magnesium (decigrams)	154	0.1	**154.1**
Copper (decigrams; incl. pipes)	550	25.5	**575.5**

Table 4.3: Per-capita material masses in Oslo's water and wastewater pipeline network as at end of year-2006.

[1] Ductile iron alloy composition: C3.35, Si2.5, Mn0.35, S0.01, P0.02, Mg0.04, Fe93.73
[2] Grey cast iron alloy composition: C2, Si2, Fe96
[3] Mild steel alloy composition: C0.2, Si0.3, Mn1, S0.05, Cu0.3, Fe98.5
The compositions are averages assumed to be constant over time, and sourced from the respective Wikipedia pages

It is seen that grey cast iron accounted for 64% of the total mass of materials of construction of functional water pipelines, while the corresponding material for wastewater pipelines was concrete at 93%. Ductile iron was at the number-two position in the case of water pipelines, while plastics occupied the corresponding spot for wastewater pipelines. All materials of construction in the pipeline networks - taken together - had a mass of around 437,000 tonnes (approximately 764 kilograms per capita), with water pipelines accounting for 21 % of the total. Iron metal in the pipeline networks (including the reinforcements in concrete pipes) accounted for 23% of the total mass. It should be repeated again at this juncture that these masses pertain to only the 'operational' or 'active' component of the pipeline networks. Data about the retired pipelines – or obsolete stock – were not available.

When the per-capita mass of pipeline materials of construction (for water and wastewater networks taken together) for Oslo, Trondheim and Tromsø were plotted against the population density of the respective cities, the possibility of a correlation between these two parameters is revealed. A logarithmic equation seems to be the best fit to the data.

$$y = 5630 - 687\ln(x) \qquad \text{- (Eq. 4.1)}$$

Here, y = per-capita mass and x = population density in resident population per square kilometre. As the population density increases, the per-capita mass decreases. However, if the value of x is more than 3626 persons per square kilometre (for *Equation 4.1*), y becomes negative. This equation can thus be tested for cities with population densities below this number, and all Norwegian cities would thereby qualify. It has an R^2 value equal to 0.998, but with just three data points, it is unscientific to generalise such a correlation. To verify and confirm the existence of such a correlation and thereafter the nature of the correlation, quantitative data about the pipeline networks of more cities are needed. (This possibly could be attempted as a part of further research in this area.)

Though manhole covers have been left out of the analysis, it would be apt to just mention here, on the basis of data (and information) obtained from Westheim [112] that, as at the end of year-2006, the total mass of the 45,000 manhole covers (Ugarelli et al [25]) in the Oslo network was 2100 tonnes (i.e. 46.5 kilograms per cover). As Westheim [112] has further informed, over 98% of the manhole covers are made of grey cast iron. The maximum lifetime is 15 years, and at end of life, the manhole covers are usually re-smelted. The total mass of grey cast iron in the water and wastewater pipeline networks, in year-2006, to afford a comparison with the mass of manhole covers in stock, was thirty times greater. The total mass of all pipeline materials taken together was over 210 times greater (*Table 4.3).*

Energy consumption: Referring back to *Figure 3.1*, it is seen that energy is consumed all along the life-cycle of a pipeline – from production to retirement from service. The production-phase energy consumption has not been determined separately, as this has been automatically incorporated into the LCA studies (and the embodied energy analysis as well). As mentioned in the previous chapter, the energy consumed during the final retirement phase has been deemed to be negligible, and thereby not taken into consideration. The consumption of diesel fuel during the installation, rehabilitation and operation and maintenance (O&M) phases, for the period 1991-2006, is depicted graphically in *Figure 4.8* and *Figure 4.9.* The scales on the Y-axes for the two graphs are the same, for the sake of easy comparison. In the water pipeline network, till year-2002, the installation phase accounted for the bulk of the diesel consumption (between 79% and 87%). From year-2003 to year-2006, it was the O&M phase which accounted for almost all of the consumption (between 84% and 99.5%). During the 16-year period, approximately 4.73 million litres of diesel – annual average of 295,000 litres - were consumed in the three referred-to phases of the water pipeline network. In energy terms (using a density of 0.8 kg/litre and a calorific value of 45 MJ/kg), this is equivalent to 170,615 GJ or 47 GWh. That gives an annual average of around 3 GWh.

For the wastewater pipeline network, the installation phase dominated all through till the end of the time period referred to, though the share of the O&M phase consumption in the total increased over time – from 6.3% in 1991 to over 35% in 2006. Vis-à-vis the 4.73 million litres consumed in the water pipeline network, 7.7 million litres (annual average of 481,200 litres) were consumed in the wastewater pipeline network. In energy equivalents, that would be 277.3 TJ or 77 GWh, giving an average of 4.8 GWh per year.

Thus, both networks taken together, consumed 12.43 million litres of diesel (equivalent to 447.9 TJ or 124 GWh or 7.8 GWh on average per year) in 16 years, in their installation, rehabilitation and O&M phases. The consumption by the wastewater pipeline network decreased almost steadily from 1 million litres in year-1991 to 183,000 litres in year-2006 (*Figure 4.9*), while the consumption by the water pipeline network oscillated around a mean of about 376,100 litres from year-1991 to year-2002, before plummeting suddenly to an average of 56,500 litres for the next four years (*Figure 4.8).* The embodied energy in the functional pipelines in both the networks has been calculated using the values tabulated in *Appendix 5.* The scope, as enunciated earlier, has been restricted to pipeline material production and fabrication only. As the purpose was to compare the networks to those in Trondheim and Tromsø for which the databases were updated till year-2008, it has been assumed on the basis of conversations with Kristiansen [16], that the additions to the pipeline networks in Oslo in 2007 and 2008 have been negligible. (This would then mean that the mass of pipeline materials in the active stock at the end of year-2008 was not very different from the same, at the end of year-2006.) The embodied energy in the wastewater pipeline network and the water pipeline network in Oslo were 3446 MJ per capita (or 547 GWh) and 5944 MJ per capita (or 945 GWh) respectively. The predominance of concrete in the wastewater pipeline network and ferrous materials in the water pipeline network explains the gap between the two values. The energy consumed during the installation phase of *all* the active pipelines in both the networks taken together, at 1715 GWh, was 1.15 times greater than that embodied during the production and fabrication phases. The embodied energy for Trondheim and Tromsø, vis-à-vis 1492 GWh in Oslo, were 1918 GWh and 198 GWh respectively at the end of year-2008.

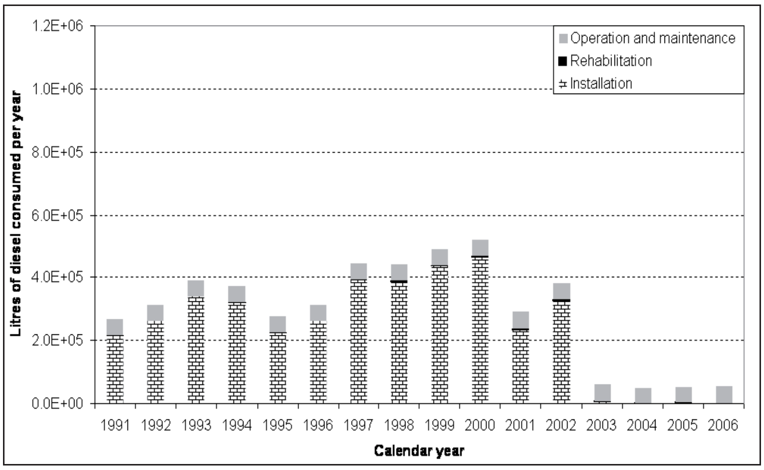

Figure 4.8: Diesel fuel consumption in the water pipeline network in Oslo (1991-2006)

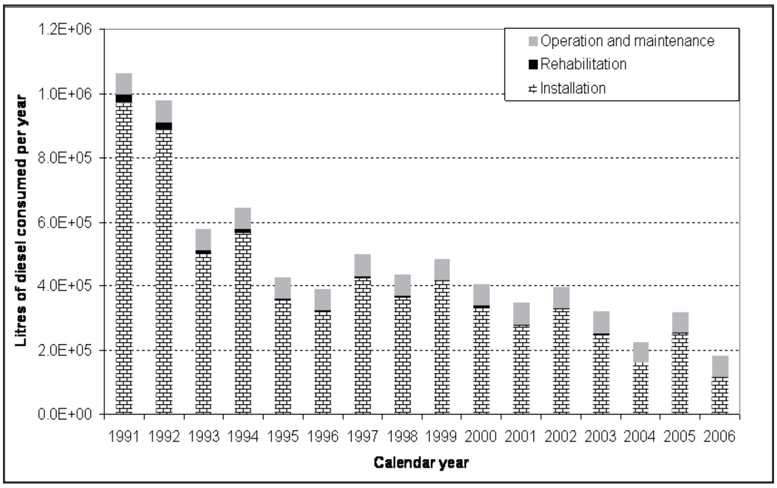

Figure 4.9: Diesel fuel consumption in the wastewater pipeline network in Oslo (1991-2006)

Optimising expenditure: The annual costs of managing pipeline networks are closely related to the physical pipeline stock characteristics – length, number, material composition and age distribution in the network. In pipeline networks which are nearing saturation, the annual expenditure can be bifurcated into two components – rehabilitation investments and O&M expenses. Network expansion investments, if any, are negligible in comparison. If the time period 2004-2008 is considered, it is seen that the ratio of investments made in the wastewater transport system to those made in the water distribution system increased from 0.85 in year-2004 to 1.32 in year-2008 [116].

In Ugarelli et al [25] and Venkatesh et al [120], the LCC method (Fuller [44]) has been employed for a forecast period of 20 years (years 2008-2027), to prove that the physical lifetime approach to the rehabilitation of wastewater pipelines, can yield better all-round end-results, vis-à-vis the in-vogue economic lifetime approach. For instance, if no additions at all are made to the wastewater pipeline network in the forecast period, and the physical lifetime approach is adopted in preference to the economic lifetime approach, the annual rehabilitation investment called for would be less by about 5.8 million year-2008-€, and the total expenses would decrease – by around 88.1 million year-2008-€. The ageing phenomenon of the pipeline network can also be effectively tackled – in the said case, with the economic lifetime approach, the average age of the network would be 47 years at the end of year-2027, while with the physical lifetime approach it will be 12 years younger in comparison. The capital value of the rehabilitated pipelines is assumed in both cases to be depreciated at the rate of 2% per year. However, as advocated by Ugarelli et al [25], while the physical lifetime approach is certainly superior to the economic lifetime approach, condition monitoring cannot be ruled out. Ideally, replacing the right pipe at the right time should be the goal. This has also been emphasised in Ugarelli et al [130]. Owing to data non-availability at the time this research was done, a similar LCC for the water pipeline network could not be carried out.

Environmental impacts: In Venkatesh et al [121] and Venkatesh and Brattebø [122], the environmental impacts associated with the water and wastewater pipeline networks have been analysed. *Figure 4.10* depicts the aggregated life-cycle environmental impact scores after normalisation and weighting for the water and wastewater pipeline networks respectively.

Crushed gravel used for the bedding material may be significant in terms of the mass inflows (as deduced earlier in this section), but its contribution to the aggregated environmental impact score, on average, was less than 2%. The scores do not include the contribution of the bedding material (crushed gravel). The impacts attributable to the water pipeline networks were greater than those attributable to the wastewater pipeline networks in the years 1995, 1996, 1997, 1999, 2000, 2001 and 2002. In the other years, it was the converse. Overall, one may conclude that both the sub-systems had more or less an equal and decreasing environmental impact. For both the pipeline networks, the global warming potential and the abiotic depletion potential were the two major environmental impact categories, with the production-installation phase contributing the most. In the case of water pipeline networks, the medium-diameter pipelines were the biggest contributors to the impacts in the said period. It should be mentioned at this juncture that the impacts can also be expressed in terms of the specific aggregated impact scores – per metre of active, operational pipeline in the network. In that case, the decrease over the time period 1991 to 2006 would be more conspicuous than what is borne out in *Figure 4.10,* owing to a progressively-increasing denominator-term.

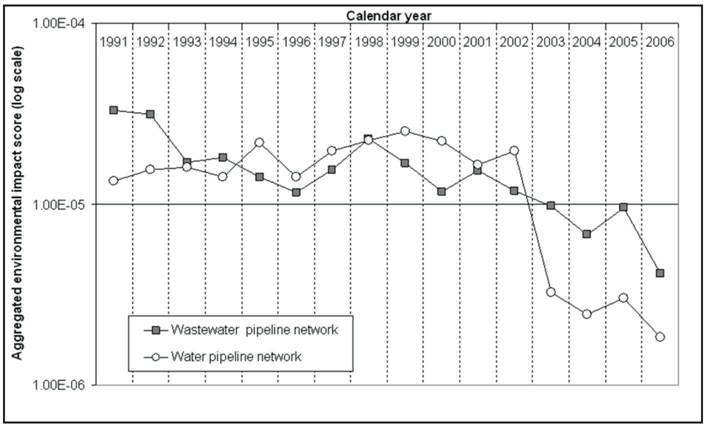

Figure 4.10: Aggregated environmental impact score for water and wastewater pipelines in Oslo (1991-2006)

In Venkatesh et al [121], it has been shown that by adopting the physical lifetime approach advocated in Ugarelli et al [25] and Venkatesh et al [120] *in lieu* of the economic lifetime approach, during the forecast period of 2008-2027 considered in the papers referred to, the annual greenhouse gas (GHG) emissions from the wastewater pipeline network, could be decreased by 150 tonnes of CO_2-equivalents. With both networks almost totally saturated at the time of writing, the production-installation phase will become almost non-existent in the forecast period, decreasing the total environmental impacts considerably. In 1991, the GHG emissions amounted to 5,000 tonnes of CO_2-equivalents. In 2027, even in the worst-case scenario (one of the seven considered in Venkatesh et al [121]), it will be well below 1,700 tonnes of CO_2-equivalents. The corresponding environmental impact scores are 3.31E-5 (from *Figure 4.10*) and 1.16E-5.

With rehabilitation becoming extremely crucial for ageing pipeline networks like Oslo's, which are almost saturated, as PU enables dematerialization (by a factor of 2.4) and thus decreases the aggregated environmental impact score by a factor of 3.4 (see *Appendix 7),* it may be an attractive substitute for epoxy resin in the future.

Pipeline blockage analysis: Blockages of wastewater pipelines occur owing to the interplay of a host of factors – both inherent in the pipe and external to it. Basement flooding could be a consequence when the blocked pipe fails. Good asset management includes the prevention of blockages or attending to them before basement flooding occurs. This necessitates an understanding of the entire network and the proneness of its constituents to blockages.

Function	**Pipes**	**Blocks**	**Block rate**	**Block factor**
Combined flow	1,291	177	0.146	1.14
Stormwater	685	2	0.003	0.03
Sewage	1,734	265	0.176	1.37
Material	**Pipes**	**Blocks**	**Block rate**	**Block factor**
Concrete	1,319	293	0.228	1.78
Plastics	2,391	151	0.075	0.58
Age in years	**Pipes**	**Blocks**	**Block rate**	**Block factor**
0-19	1,478	52	0.043	0.34
20-39	1,375	142	0.111	0.91
40-59	456	113	0.278	2.16
60-79	284	87	0.259	2.02
80-99	87	31	0.302	2.35
100-116	30	19	0.638	4.96
Diameter (mm)	**Pipes**	**Blocks**	**Block rate**	**Block factor**
100-140	24	0	0	0
150-180	283	83	0.364	2.83
200-230	1,930	286	0.160	1.24
250-280	407	24	0.071	0.55
300-335	741	48	0.076	0.59
350-380	88	3	0.040	0.31
381 - 400	237	0	0	0
Slope in ‰	**Pipes**	**Blocks**	**Block rate**	**Block factor**
0-15	1,239	243	0.205	1.59
15-30	719	72	0.114	0.88
30-45	501	30	0.071	0.55
45-60	387	39	0.110	0.85
60-75	306	26	0.094	0.73
75-90	227	11	0.061	0.47
90-105	151	8	0.072	0.56
105-120	98	8	0.112	0.87
120-135	59	5	0.107	0.83
135-140	23	2	0.121	0.94

Table 4.4: Pipeline blockage analysis results for Oslo's wastewater pipeline network (1991 to 2006) (Block rate is in blockages per kilometre per year; and Block Factor is a unitless number)

As inferred in Ugarelli et al [125], sewage pipelines are more prone to blockages than combined flow and storm-water pipelines. This possibly is owing to the fact that the sewage carries with it a host of solid impurities, is denser and more viscous and thus relatively sluggish in comparison to storm-water and mixed flow, with respect to fluidity. While this can even otherwise be deduced by intuitive reasoning, the analyses serve as a strong confirmation of the intuition. There is a distinct inverse relationship between the diameter of the pipeline and its proneness to blockage. It can also be expected that there would be an inverse relationship between the slope and the said proneness. *Table 4.4* reproduced from Ugarelli et al [125], summarises the results of the analysis. However, from the volumes of data handled for this study, this is not at once stark and clear. Probably, availability of data for the years before 1991 is an essential pre-requisite for a stronger proof of this relation. The time-period of 16 years considered for this analysis may thus be a limitation which can only be overcome if access to greater volumes of data can be made possible.

With age, it is evident that pipelines deteriorate and can more easily suffer blockages. Thus, a critical candidate would be a small-diameter, old sewage pipeline made of concrete, laid almost horizontal to the ground surface. It is difficult to estimate the relative importance of the different influencing factors, to the proneness of pipelines to blockages. The results of this analysis can be fed into a subsequent economic analysis of proactive and reactive measures that can be undertaken by Oslo VAV to optimise expenditure and maintain the level of performance, by availing of blockages-flooding data (the average compensation paid by Oslo VAV per flooding event) from this paper, and estimates of rehabilitation, and O&M expenses.

4.1.2. Pumping stations

Energy consumption: Both water and wastewater pumping stations have been dealt with, in this sub-section. As mentioned earlier, owing to lack of access to reliable data, the possibility of performing an MFA or a stock analysis of pumping stations has been ruled out. The access to water and sewage pumping energy data was limited to only some years in the period 2000-2009, and hence, on the basis of the known values and the assumption of proportionality to the volumes pumped, the consumption figures for the other years of the 2000-2009 time-period have been calculated. The energy consumption during the maintenance operations has been neglected on the grounds of negligibility vis-à-vis the operational energy consumption (Reksten [18]). The latter values have been tabulated in *Appendix 8.* The average annual energy consumption for water pumping during the 10-year period was around 16.4 GWh, while the same for sewage pumping was approximately 6 GWh. (Together, they accounted for about 17%, on an average, of the total energy consumption in the system (Venkatesh [126])). The large difference between sewage and water pumping can be partly explained by referring to the locations of Oset - the water treatment plant (WTP) that supplies 90% of the water consumed by Oslo, and BEVAS and VEAS (see *Figure 1.3).* The consumer population is primarily concentrated in the Oslo city centre, south of Oset WTP; and the mean distance between the WTP and consumer is greater vis-à-vis that between the consumer and the wastewater treatment plant (WWTP) to which he/she discharges wastewater. (Note that the WWTPs VEAS and BEVAS straddle the Oslo fjord and handle sewage from the west and east of the city respectively).

Environmental impacts and costs: The pumping stations taken together – sewage and water – accounted for around 30% of the total global warming potential of the water-wastewater system, when the electricity drawn by the pumps was assumed to be the Nordic mix (Venkatesh and Brattebø [126]) *(*Refer *Appendix 6)*. This was equivalent to an average of 7.5 kg of CO_2-eq per capita per year of the population serviced. However, the pumping stations accounted for just below 5% of the total aggregated environmental impact score associated with energy consumption in the O&M phase. The annual pumping energy costs for the period investigated were between 2 € and 3 € per capita (Venkatesh and Brattebø [126].)

4.2. Water and wastewater treatment

WTPs and WWTPs (refer *Figure 1.2*) have been dealt with separately, in two sub-sections under this section. For WTPs, the ten-year time period 2000-2009 has been studied (with years 2005 and 2006 not being considered owing to non-comprehensive data), and for WWTPs, the focus has been on the eight years 2000-2007.

4.2.1. Water treatment plants

Chemicals consumption: From year-2000 to year-2004, chlorine was used as a disinfectant at all the four WTPs, and aluminium sulphate served as the coagulant at Skullerud (*Appendix 4).* In year-2007, sodium hypochlorite made an entry, and chlorine was phased out entirely in year-2008. Likewise, in year-2008, when chemical treatment was introduced at Oset, the WTP started off with polyaluminium chloride (PAX) as a coagulant in *lieu* of aluminium sulphate. Aluminium sulphate continues to be used at Skullerud though. In year-2000, about 1,023 tonnes of chemicals were consumed in all the four plants taken together to treat a total of about 96 million m^3 of water, at a rate of around 10.7 grams per m^3 of water treated. This rose to a little over 12 grams in year-2001, before declining to 10.1 grams in year-2004. A decrease in aluminium sulphate consumption played a key role in this decline, while specific consumption of chlorine and polymer increased slightly, and that of carbon dioxide and calcium hydroxide almost remained constant. The optimization of the use of aluminium sulphate may have been occasioned by the concern over the presence of excess aluminium in the treated water, which has been, as mentioned in Droste [64], believed to cause Alzheimer's disease. The increase in the use of chlorine for disinfection can be attributed to consumer health concerns. The startling jump in chemicals consumption from 1,174 tonnes to 5,119 tons (in year-2008) to 9,547 tonnes (in year-2009) is easily explained by the fact that in year-2008, about 40 million m^3 of water were treated chemically at the Oset WTP for the first time, and in year-2009, all the inflow at Oset received chemical treatment. The specific consumption thereby rose from 12.3 grams in year-2007 to 53.3 grams in year-2008 and 97.4 grams in year-2009. New entrants into the chemicals mix in year-2008 were sodium hypochlorite which replaced chlorine entirely at all the plants, and PAX and microsand at Oset.

There are mass outflows to the hydrosphere (treated water) and the atmosphere, related to the consumption of chemicals at the WTPs. However, owing to non-availability of accurate data, these have not been estimated. The environmental impacts associated with these outflows have also thereby not been measured. The sludge from the WTPs is the sink thereby, for the treatment chemicals as well as the constituents separated from the raw water. From 920 tonnes in year-2000, the mass of dry solids in the WTP sludge increased over ninefold, to 8,590 tonnes in year-2009.

Energy consumption: The electricity consumed rose from 20.0 GWh (equivalent to 0.2 kWh per m^3) in year-2000 to 33.6 GWh (0.34 kWh per m^3) in year-2009 (*Appendix 4*). From year-2004 onwards, power supply interruptions had to be balanced by consuming diesel fuel in in-plant generators at the Skullerud WTP. The mass of diesel consumed increased from 151.9 tonnes in year-2004 to 191.4 tonnes in year-2009. If expressed in terms of the total water treated by all the four WTPs (though diesel fuel consumption was restricted to the Skullerud WTP alone), the corresponding numbers were 1.63 grams and 1.95 grams per m^3. The atmospheric outflows associated with diesel consumption at Skullerud, are factored into the LCA; and not specified separately. Likewise, the inflow of air (oxygen) – a vital mass input - for the combustion of diesel at the plant has been ignored.

Costs and cost indicators: When expressed in year-2008 currency units, the annual investments in WTPs in Oslo, during the 2004-2008 time period, peaked in year-2006 to 21.1 million €. The average annual investment during the said period was around 16 million €. The annual investments in WTPs were less than the corresponding ones in the water transport system, in years 2004, 2005 and 2008. In years 2006 and 2007, it was the other way round, with the ratio of the former to the latter being 1.93 and 1.42 respectively (calculated on the basis of data sourced from Statistics Norway [116]). While capital investments (CAPEX) are needed to upgrade and expand the water treatment sub-system, and equip it with newer and state-of-the-art machinery and equipments, the operational expenses (OPEX) are more run-of-the-mill, and are needed to keep the WTPs functioning on a daily basis. The OPEX, *inter alia,* includes the expenditure on chemicals and energy consumed at the WTPs.

While the volume of water treated and supplied did not vary much over the time-period 2000-2009 (ranged between 92.8 and 98 million m^3 annually), the total mass of chemicals consumed, increased nine-fold. All the costs in the discussion that follows are expressed in year-2009-€. The total expenditure on chemicals rose from 0.24 million € (around 0.25 €-cents per m^3) in year- 2000 to 1.51

million € in year-2009 (1.55 €-cents per m^3). The expenditure on energy was between 4 to 6 times greater than that on chemicals in the period 2000-2004. *Figure 4.11* depicts the changes in the relationship between the specific costs indicators for chemicals and energy, for the time period considered. Unit costs are tabulated in *Appendix 4.*

About 33 GWh of electricity was consumed, and diesel consumption increased slightly to 2.31 GWh. Owing to the compounded effect of a rise in tariff (29%) and a rise in consumption (50%) necessitated by the energy requirements (essentially for mixing) of the chemical treatment processes at the Oset WTP, the energy cost per m^3 of water treated, increased. As more chemicals were introduced into the system to improve the degree of treatment, the expenditure on chemicals increased at a much faster rate, vis-à-vis that on energy – it sextupled from 0.25 €-cents in year-2007 to over 1.5 €-cents per m^3 in year-2009. In the years 2000, 2002, 2003 and 2004, chlorine gas accounted for the largest portion of the chemicals-costs-pie (varying between 32% and 38%). In year-2001, it was aluminium sulphate (28%), in year-2007, sodium hypochlorite (43%); and in the years 2008 and 2009, PAX (29% and 35% respectively) (Venkatesh and Brattebø [127]).

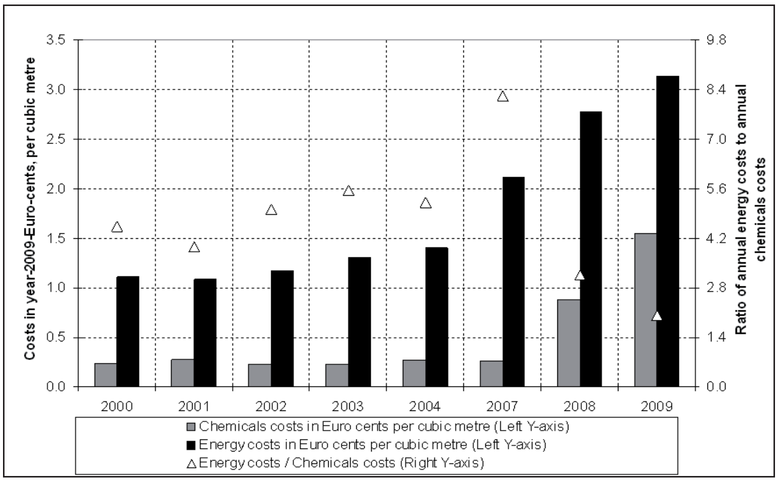

Figure 4.11: Specific chemicals and energy costs at Oslo's WTPs (2000-2009; years 2005 and 2006 not shown)

Environmental impacts: The environmental impacts in this analysis are associated with the production and transport of chemicals, generation and transmission of electricity, production and use of diesel in generators and the transport of sludge to WWTPs for final treatment. *Figure 4.12* compares the specific (per unit volume water treated) aggregated environmental impact scores for chemicals and energy consumption using the Norwegian electricity mix (see *Appendix 6).* In year-2009, the ratio of the specific score for chemicals to that for energy stood at 4.8. The impacts due to chemicals consumption rose ten-fold from year-2000 to year-2009, owing to the previously-mentioned introduction of upgraded chemical treatment at the Oset WTP. The impacts due to energy consumption rose nine-fold during the same period. Global warming dominated the impact score for both energy and chemicals consumption. Acidification was significant in the case of energy consumption, owing to the diesel component of the energy mix.

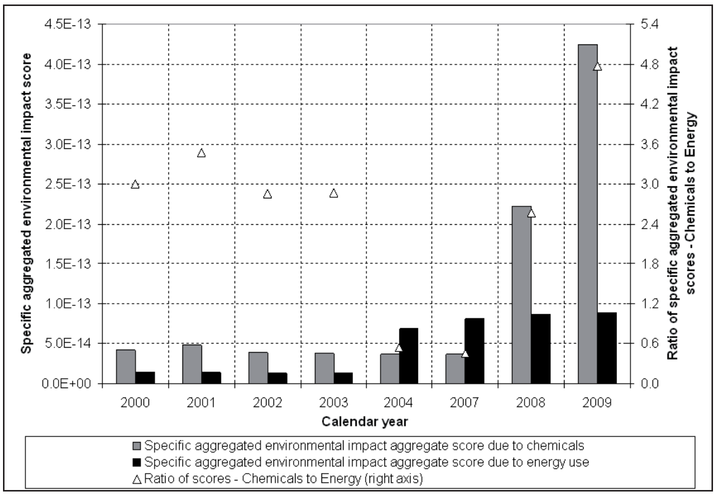

Figure 4.12: Specific aggregated environmental impact scores for chemicals and energy at Oslo's WTPs (2000-2009; Norwegian electricity mix used; and years 2005 and 2006 not shown)

Calcium hydroxide was the biggest contributor (inferred on the basis of the impact scores) in years 2000 (32.1%), 2003 (35.0%) and 2004 (34.5%), carbon dioxide dominated in years 2001 (33.9%), 2002 (35.1%) and 2007 (32.9%). In years 2008 and 2009, PAX contributed the most with 33.7% and 38.2% respectively. Referring back, it is seen that PAX also accounted for the biggest share of the costs of chemicals in the years 2008 and 2009. The impacts caused by transportation – barge and road combined – varied between 4.5% and 10% of the total. The share was the highest in year-2000 at 9.7%. From less than 10 grams of CO_2-equivalents per m^3 of water treated in year-2000, the GHG emissions attributable to chemicals and energy consumption rose to nearly 85 grams of CO_2-equivalents in year-2009 (Venkatesh and Brattebø [123]).

If the Nordic electricity mix (see *Appendix 6*) is adopted instead of the Norwegian mix, for the LCA, stark differences are observed. The impacts attributed to chemicals consumption, would then increase slightly - by between 4% and 10%; while the impacts associated with energy consumption would increase drastically – the range being 81% to 420% (Venkatesh and Brattebø

[123]). The difference, evidently, is owing to the higher fossil fuel content of the Nordic mix.

4.2.2. Wastewater treatment plants

Chemicals consumption: *Appendix 4* lists all the values associated with chemicals consumption and costs. In year-2000, 22,500 tonnes of chemicals were consumed in the WWTPs in Oslo (that equates to around 189 grams per m^3 of wastewater treated). Accounting for the bulk of the consumption was the coagulant iron chloride. When the much-stronger iron sulphate (according to Droste [64], iron sulphate has much greater coagulating ability for both positive and negative species, than iron chloride) was introduced in year-2001, the overall demand for iron salts dropped, and iron chloride consumption decreased by 60%. In specific consumption units, there was a reduction in chemicals consumption from 189 grams per m^3 of wastewater treated in year-2000 to 131 grams in year-2007. The specific consumption of PAX varied between 18.11 grams to 36.5 grams per m^3 of wastewater treated. Methanol consumption varied in a relatively narrower range between 17.4 grams and 21.7 grams per m^3. This was true for calcium carbonate (17.6 grams to 23 grams per m^3) and methanol (11.7 grams to 16.9 grams per m^3). Ethanol consumption peaked to 19.8 tonnes in year-2005, before dropping to nil in year-2007 (Venkatesh and Brattebø [124]).

The partitioning of the chemicals consumed, into the three outflow streams – treated effluent (hydrosphere), sludge (biosphere, pedosphere and lithosphere), and atmospheric emissions – is essential for the LCA. So is the tracking of the fate of the constituents of the influent wastewater at the exit of the WWTPs. (In Venkatesh and Brattebø [128] & [129], for instance, the partitioning of the influent wastewater components has been done, on aggregated national scales, for the Dutch and the Norwegian wastewater treatment plants, respectively.) Small portions of the influent constituents and chemicals consumed, exit along with the treated effluent, and some are converted into gaseous emissions during the treatment processes (such as degradable organics in the influent, and methanol and ethanol). There is by-product recovery (ammonium nitrate formed by a combination of ammonia from the influent and the nitric acid consumed) on the downstream at the VEAS WWTP. The portion of the influent organics which is not degraded during anaerobic sludge digestion, and the fraction of the chemicals consumed which is not separated during the downstream sludge handling processes, end up in the sewage sludge destined for final end-use in agriculture, landscaping and silviculture (BEVAS [20], VEAS [22] and Venkatesh and Brattebø [129]). The mass of dry solids in the sludge leaving WWTPs fluctuated between 5,180 tonnes and 7,720 tonnes during the period 2000-2007 and the heavy-metal content of the sludge was low enough to permit its utilisation as a fertiliser substitute. These flows have been discussed in Venkatesh and Brattebø [124].

Energy consumption: *Appendix 4* lists all the values associated with energy consumption. *Figure 4.13* presents a time-series of the heat and electricity consumption in the VEAS and BEVAS WWTPs. The electricity purchased from the grid remained fairly constant at around 28 GWh to 30 GWh, while that produced in-house from the biogas rose steadily. In year-2000, the in-house production accounted for 22.9% of the total electricity consumed; and in year-2007, this increased slightly to 24.3%. On a per-cubic-metre-of-wastewater-treated basis, the total energy consumption rose from 0.61 kWh to 0.78 kWh (by 27%), the average for the 8-year period being 0.75 kWh. It must be mentioned that the consumption of heat energy as indicated in *Figure 4.13*, is not the same as the actual heat demand at the plants. It follows that there are heat losses during use, to the ambience, effluent wastewater and the sludge (with an attractive unharnessed recovery potential, as described for the Japanese case study in Funamizu et al [76]).

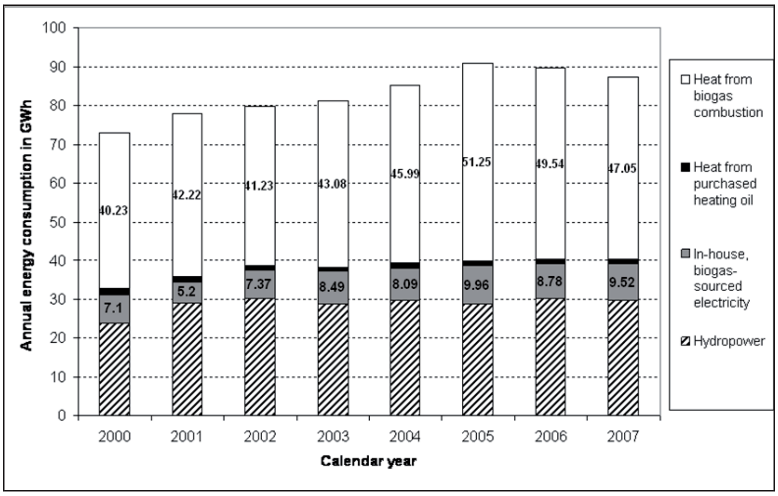

Figure 4.13: Energy consumption in Oslo's WWTPs (2000-2007)

There was a 34% rise in the in-house generation of electricity from biogas, from 7.1 GWh to 9.5 GWh, while the volume of biogas generated (volume measured at normal temperature and pressure conditions) rose by over 70%, from 8.1 million m^3 to 14 million m^3. At present, it is only at VEAS that electricity is generated using the biogas produced in-house. With a rise in the efficiency of conversion – heat to electricity – it is quite possible that in the future, the electricity yield could be improved.

Costs and cost indicators: When expressed in year-2008 currency units, the annual investments in WWTPs in Oslo, during the 2004-2008 time period, dropped from 0.73 million € in year-2004 to 0.25 million € in year-2008. The average annual investment during the said period was 0.37 million €. The annual investment in the WWTPs was between 23 and 103 times less than the corresponding investment in the wastewater transport system (calculated on the basis of data sourced from Statistics Norway [116]).

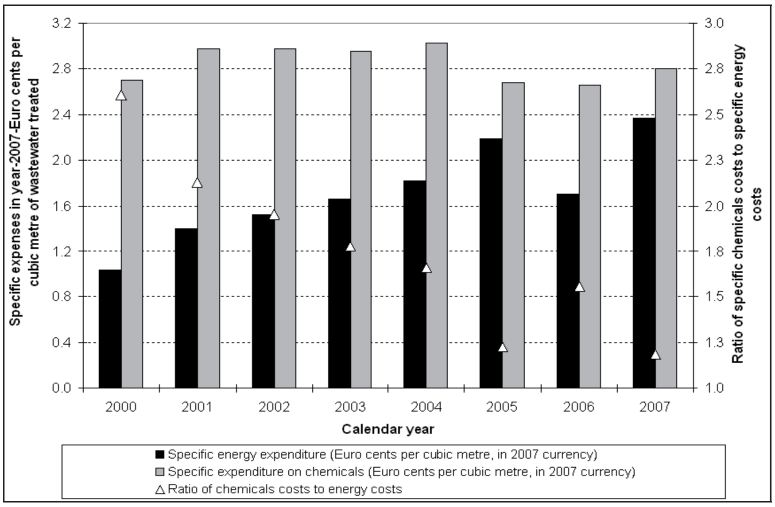

Figure 4.14: Specific expenses on energy and chemicals at Oslo's WWTPs (2000-2007)

When the expenses on chemicals are calculated in terms of year-2007-€, it is seen that they varied between 2.98 million € and 3.32 million €. The expenditure on energy includes only the costs of fuel oil and electricity from the grid; and not the cost of generating biogas and electricity in-plant. Hence, one may say that the energy expenditure is understated. When compared to energy, chemicals were pricier throughout the seven-year period. From around 2.7 €-cents in year-2000, the specific expenditure on chemicals reached a peak of more than 3 €-cents in year-2004, and then decreased (*Figure 4.14*). Energy expenses rose from a value of around 1 €-cent in year-2000 to 2.4 €-cents in year-2007. The ratio of the specific expenditures dropped from 2.6 to 1.2 showing an almost-steady decrease, save the fluctuation between the years 2005 and 2006, occasioned by a decrease in electricity prices (Venkatesh and Brattebø [127]).

Environmental impacts: The major environmental impact category due to energy consumption was acidification, courtesy the emissions of NO_x (nitrogen oxides) and sulphur dioxide from biogas combustion. Owing to the generation and consumption of more biogas over time (as mentioned earlier, the volume of biogas consumed increased by over 70% during the 8-year period), the specific aggregated environmental impact score for energy consumption (per cubic metre of wastewater treated) nearly doubled between the years 2000 and 2007. What is noteworthy besides the domination of acidification is the negative abiotic depletion and negative global warming potentials. This can be explained by the role of biogas in obviating the need for production of an equivalent amount of natural gas.

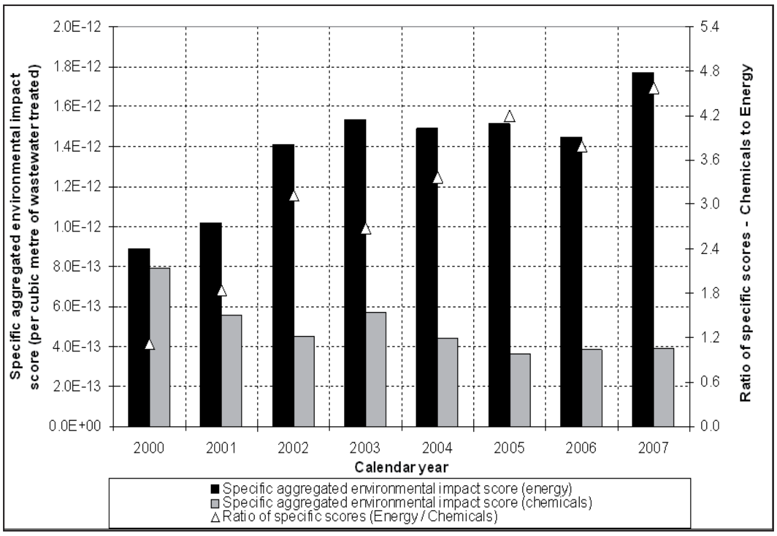

Figure 4.15: Specific aggregated environmental impact scores for energy and chemicals consumption at Oslo's WWTPs (2000-2007; Norwegian electricity mix used)

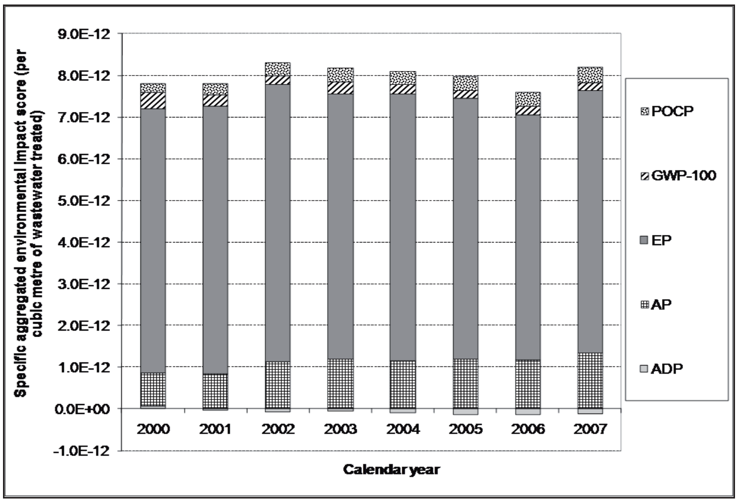

Figure 4.16: Specific aggregated environmental impact scores for the operation phase of WWTPs in Oslo – chemicals and energy use and treated effluent wastewater (2000-2007; Norwegian electricity mix)

In the case of chemicals, the major impact category was global warming, followed by abiotic depletion. Interestingly, it is these two impacts which had a subtractive effect on the score in the case of energy. The avoided production of urea, superphosphate and ammonium nitrate brought down the impacts quite significantly (by over 50% in most cases). When a substantial portion of the highly-impacting coagulant ferric chloride was replaced by ferric sulphate in year-2001, the impacts decreased conspicuously. The carbon dioxide originating from the ethanol and methanol used as carbon sources in the denitrification process was a significant contributor to global warming. In fact, it accounted for between 10% and 25% of the aggregated score. On the premise that the transportation distances for the chemicals – from factory to plant - may not be greater than 150 kilometres, it can be shown that the contribution of transportation to the life-cycle environmental impacts attributed to chemicals was minimal. By the same token, one can also now justify the omission of the out-of-WWTP sludge transportation process.

From *Figure 4.15,* it is seen that the ratio of the specific aggregated environmental impact score for energy to that for chemicals increased over time, from close to one in year-2000, to over 4.5 in year-2007. The GHG emissions per m^3 of wastewater treated, decreased from 1.9 kilograms of CO_2-eq to 1 kg of CO_2-eq in the case of chemicals. In absolute terms, the corresponding values are 230,000 tonnes and 113,000 tonnes (a decrease of more than 50 per cent). For energy, the GHG savings increased from 2.7 grams to 5.95 grams of CO_2-eq per m^3 of wastewater treated. In absolute terms, the corresponding specific values were 330 tonnes and 663 tonnes (a doubling in 8 years).

It is the treated effluent wastewater that accounted for (and usually always does) the largest chunk of the total impacts - between 73% and 80% (also proved by Lassaux et al [82] for a Belgian case study). While acidification dominated in the case of energy and global warming in the case of chemicals, eutrophication was the key impact category when it came to the treated effluent. Eutrophication thus emerged as the major impact category in the WWTPs' operation phase, with acidification coming a distant second (Venkatesh and Brattebø [124]). The avoided production of fertilisers and natural gas introduced a net negative abiotic depletion potential in the latter years of the study period (*Figure 4.16).*

When the Nordic electricity mix is considered, there are no appreciable changes in the environmental impacts due to chemicals consumption. The impacts attributable to energy increase slightly with the Nordic mix – over a range of 2% to 7%. While there are savings in GHG emissions (negative GWP-100 potential in other words) with Norwegian electricity throughout the study period, net positive emissions result when the Nordic mix is used, for all years except year-2005. The increase in specific GHG emissions with the Nordic electricity mix vis-à-vis the Norwegian electricity mix is between 92% (in 2005) and 320% (in 2000). Further, when the Nordic mix is used, the percentage shares of energy and chemicals in the overall operation-phase impact score, increase, vis-à-vis the Norwegian mix (Venkatesh and Brattebø [124]).

4.3. Overall system

Figure 1.4 presents a schematic sketch of the components of, and the flows within, to and from an urban water and wastewater system. The energy consumption in the O&M phase of all the sub-systems, and the environmental impacts thereof have been studied in Venkatesh and Brattebø [126]. In this section, the energy analysis for the entire system is discussed in brief first and is followed by the environmental assessment.

4.3.1. Energy consumption in O&M phase

Figure 4.17 shows the per-capita energy consumption values for each of the six components (sub-systems) of the system. For the seven-year period, the average was 241 kWh/cap/year. Of this, water and wastewater treatment accounted for 81.6%, with the latter accounting for over four-fifths; water and sewage pumping accounted for 17.2%, with the former accounting for nearly three-fourth; and the diesel energy expended on pipeline rehabilitation and maintenance made up the rest. In absolute terms, an average of 126.5 GWh of energy (57% electricity and 43% heat from biogas combustion and heating oil) were consumed every year in the study period. The chemical energy provided by diesel for the rehabilitation and maintenance of the pipeline networks was negligible in comparison.

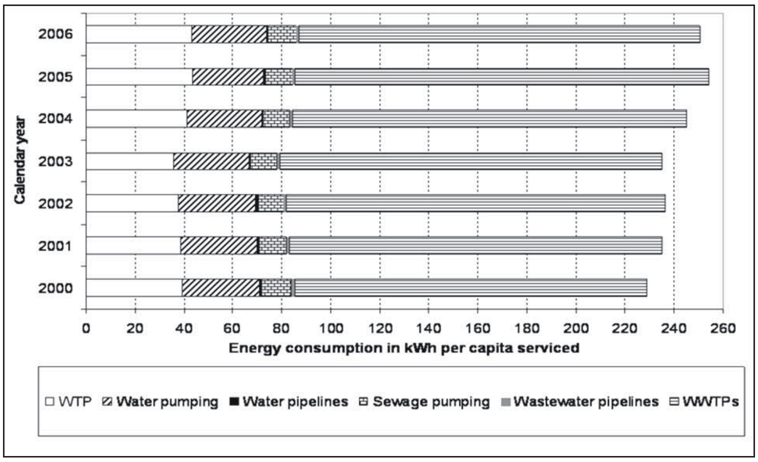

Figure 4.17: Per-capita energy consumption in the O&M phase of the water and wastewater system in Oslo (2000-2006)

Table 4.5 compares the per-unit-volume energy consumption for the upstream and downstream of the system. It is seen that the wastewater collection and treatment sub-system consumed nearly twice as much energy per unit volume wastewater handled, as compared to the water treatment and distribution sub-system per unit volume water supplied.

Calendar year	Water treatment and supply	Wastewater collection and treatment
	kWh/m^3	
2000	0.39	0.67
2001	0.39	0.76
2002	0.38	0.84
2003	0.38	0.83
2004	0.41	0.84
2005	0.42	0.87
2006	0.44	0.81

Table 4.5: Specific energy consumption on the upstream and downstream of the water and wastewater system in Oslo (2000-2006)

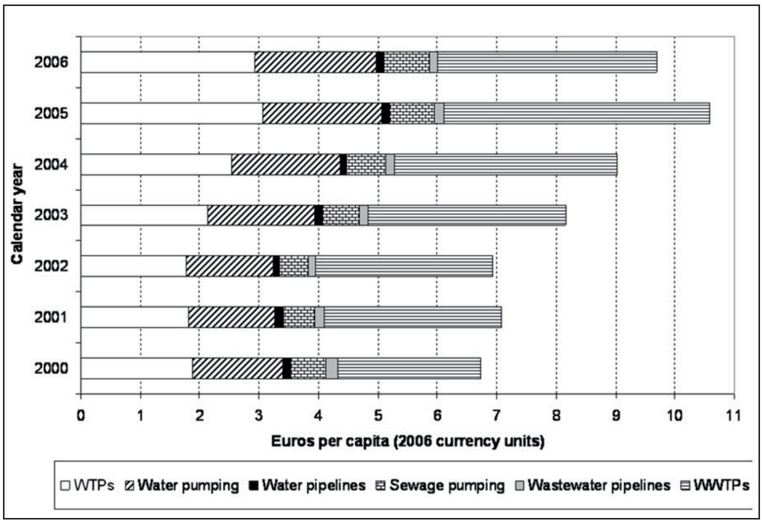

Figure 4.18: Per-capita energy costs in the O&M phase of the water and wastewater system in Oslo (2000-2006)

As a great bulk of the paid-for energy is electricity from the grid, the costs per-capita (expressed in year-2006-€) were sensitive to the electricity prices. The electricity tariff increased from year-2000 to year-2005, along with the GWh of electricity consumed (Venkatesh and Brattebø [126]). The tariff dropped in year-2006 with respect to year-2005, by 18.7%, and the effect of this is borne out clearly in *Figure 4.18.* In year-2006, 3% more electricity was purchased from the grid vis-à-vis year-2005. But the drop in the electricity price more than compensated for this increase in demand, bringing down the costs incurred by the utility. There was almost an equal split between water treatment and supply (upstream) and wastewater collection and treatment (downstream), as far as the O&M energy costs are concerned, over the 7-year period. The energy costs for the upstream were slightly greater in the years 2000 and 2006, while those for the downstream were greater in the years 2001, 2002 and 2005. The treatment energy costs accounted for between 64% and 71% of the total, pumping energy costs for between 26% and 31%, and the cost of diesel consumed in the rehabilitation and maintenance of pipelines between 3% and 5%.

4.3.2. Environmental assessment of energy consumption

In Venkatesh and Brattebø [126], the Nordic electricity mix was considered in *lieu* of the Norwegian mix. The normalized, weighted and aggregated impact scores per-capita-serviced, for each of the components of the system, for each of the seven years of the time period considered, are depicted on a log scale in *Figure 4.19.*

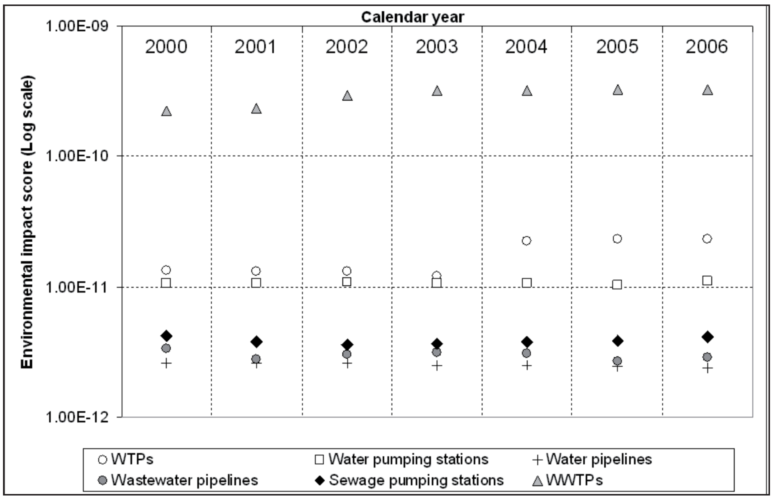

Figure 4.19: Per-capita aggregated environmental impact score attributed to energy consumption in the O&M phase of the water and wastewater system in Oslo (2000-2006; Nordic electricity mix)

The WWTPs contributed the most. The share ranged from 85.9% to 90.6% (an average of 88.2%). This was despite the fact that biogenic carbon dioxide was neglected, complete combustion of methane was assumed, and the avoided production of natural gas thanks to its substitution by biogas was considered. The reason is the much-higher grid-electricity consumption at the WWTPs vis-à-vis the WTPs (about 60% more), and the acidification impact owing to the sulphur dioxide and NO_x emissions due to the combustion of biogas. WTPs came a distant second with an average of 5.4%, followed in the third place by the water pumping stations (an average of 3.45%). Over the seven-year period, the impact score increased by 47.9%. While the impact scores for the components downstream of the WTPs either remained steady or showed a gradual increase/decrease, the contribution of the WTPs shot up suddenly from 3.4 % of the total in year-2004 to 6.2 % in year-2005; with the per-capita score almost doubling in a year. This was courtesy the use of diesel at the Skullerud WTP to compensate for the grid power supply disruptions.

For the upstream, the specific impact score increased from 1.44E-13 to 2.15E-13 - an increase of about 48%; while for the downstream, the corresponding indicator values were 9.16E-13 and 1.48E-12 – an increase of 61%. The downstream indicators are, on an average, 8.1 times greater than their upstream counterparts.

5. Discussion

In this chapter, the research questions raised in *Chapter 1* are revisited and answered on the basis of the analyses carried out. Thereafter, the limitations of this research are outlined, followed by an articulation of recommendations for further work

5.1. Research Question 1

The question is recalled here again:

> Material stock and flow analysis (MFA), energy analysis (EA) and environmental life-cycle assessment (LCA) are key industrial ecology tools which enable an understanding of the historical metabolism of urban water and wastewater systems; and also forecast the flows and environmental impacts for the future. Sustainability indicators are increasingly becoming popular as performance measurement tools for a variety of systems, but their usefulness to urban water systems has been a topic of interest only of late.
>
> a) In what way does the use of the MFA, EA and LCA methods contribute to an in-depth understanding of an urban water system's physical composition, metabolism and environmental performance - today and over time?
> b) How effective are these methods, and what are their limitations, as regards issues such as complexity of analysis, availability of data, and robustness of conclusions?
> c) Is it possible to simplify the performance analysis of urban water systems, using selected sustainability indicators or indices, without compromising the system complexity?

Answer to (a): The objectives depicted figuratively in *Figure 1.5* include an understanding of the physical system and its metabolism, and analysing system performance on the basis of the said understanding with the end-goal of advising and supporting administrative decision-making for improvements in the future. The physical system refers to the durable capital-goods stock in the urban water and wastewater system – pipelines and associated transport/distribution components, pumping stations and treatment plants with their machinery and equipments. These are associated with material inflows with long residence times. In effect, the physical system *per se* is understood by analysing the low-frequency material flows. The low frequency flows – which can also be dubbed as infrastructure stocks - are essentially a consequence of the capital investments made by the utility into the system. These stock elements have their own specific functional lifetimes, at the end of which, they exit the system as wastes or reusable/recyclable goods. The stock-and-flow analysis coupled with knowledge of the lifetimes referred to, enable one to predict outflows and replacement inflows in the future (Mueller et al [61] & Brattebø et al [33]) While this facilitates capital budgeting on the one hand for the inflows, it can also lead one to think of suitable end-of-life handling approaches well in advance.

Analysing the medium-frequency and higher-frequency material and energy flows into the different sub-systems – transport, distribution and treatment – over time, and estimating the specific consumption of materials and energy enables the utility to measure the change in the efficiency of operation. The more-frequent material and energy flows are directly related to the running (operational) expenses incurred within the system. Having determined the annual inflows of materials and energy into the system, and the corresponding outflows and emissions (to the different environmental media), it becomes easy to calculate the life-cycle environmental impacts associated with these flows. In other words, an environmental life-cycle assessment translates the material and energy flows, and the associated emissions – into the environmental impacts caused by the existence and functioning of the urban water and wastewater system.

Table 3.1 summarises the applications of the IE tools to the (partial) fulfilment of the stated objectives. 'Partial' because, it is seen from the listings that none of the tools has been applied to the construction, overhauling, equipment maintenance, repair and retirement, and demolition phases of the pumping stations and water and wastewater treatment plants (WTPs and WWTPs). Expanding the scope of the analysis to include these phases / stages / components will surely provide additional information about the system.

Life-cycle costing (LCC) does not find mention in Research Question I(a). However, it needs to be said that while this tool has been applied to study only the optimisation of expenses and investment in the wastewater pipeline network (Ugarelli et al [25]); there is scope for its application to estimate the costs and benefits of overhauling, refurbishing and maintaining the water pipeline network, water and wastewater treatment and pumping.

Answer to (b): Most of the analyses carried out using industrial ecology tools are data-demanding; and they have been so, in the case of this research as well. Both the quality (Weidema et al [43]) and the quantity of data available determine how successful the analyses will be in terms of the comprehensiveness and accuracy of the results and their usefulness to decision-making. Often, assumptions, proxy data and generalisations have to be made to fill in the data gaps. Expert opinions and ballpark estimates provided by people in the know (refer *Section 3.2*) are often indispensable. However, there is a limit to the extent data gaps can be plugged in this fashion. It was for this very reason that all the aforesaid tools could not be applied to all the components of the system (*Table 3.1*). However, the successful application of a tool to one sub-system to obtain results useful for practical decision-making (Ugarelli et al [25] for example) is testimony to the fact that, with the availability of, and access to currently-missing data, the tool can be applied to other sub-systems as well.

While the methodologies by themselves are fairly easy to apply (not complex in other words), it is the nature of the system being studied which determines the degree of complexity. But when an urban water and wastewater system – complex by all means – is broken down into its component parts, the analyses become easier. It is now the scope and depth one defines that determine how complex each of the analyses is.

As far as the final deliverables of the tools are concerned, it is more often than not, a toss-up between accuracy (which is emphasized upon when one wants specific results to aid in decision-making at the functional / technical level) and comprehensiveness (when one seeks a broader overview for strategic decisions and overarching target-setting). An improvement in the degree of accuracy of the results would entail adjusting / defining the scope of the analyses (of the different tools) to the available, reliable data. In Ugarelli et al [130], the authors have stressed on the indispensability of a robust data collection and retrieval system with well-structured and integrated information for pipeline asset management at Oslo VAV; and concluded that data-driven decision-making should become the trend in the future when data collection will be systematized and unreliability of data will cease to be a hindrance. Simply put, the robustness of the results obtained by applying the tools depends directly on the quantity and quality (reliability, specificity, etc.) of the data available for the analyses.

Answer to (c): The scope for simplification of the performance analysis will hinge strongly on what the utilities deem to be crucial for sustainability. Indicators can be defined and measured under four broad criteria – social, economic, functional and environmental. Adoption of the MFA, EA and LCA tools to study stocks, flows and impacts over time, results in, *inter alia,* deliverables which can serve as functional and environmental indicators. The LCC tool and simple economic analysis (see *Table 3.1*) yield useful economic indicators.

For the given case – the city of Oslo - water scarcity will never be a concern. Likewise, there may not be a perceptible lack of funds for maintenance, rehabilitation and upgrading (Kristiansen [16]), and the outreach of the water and wastewater pipeline networks to the city's population (a social indicator) will always be close to 100%. As these non-issues are excluded one by one, the pool of indicators shrinks to a more-manageable size. The scope for improvements in some parts of the system may be limited. This may be because of systemic lock-ins which may be too expensive to mend or owing to the fact that the best-available, state-of-the-art methods/technologies are already being adopted. The aforesaid lock-ins also include the dependence of the utility on upstream and downstream players whose operations it cannot influence in any way (the power plant supplying electricity to the concrete producer or the steelmaker selling mild steel to the pipe fabricator from which the utility sources its pipes, for example).

The focus can then be narrowed down to those performance aspects the utility has direct control over. These can be measured and/or calculated and/or derived from primary measurements and expressed as indicators or metrics. The choice of these indicators would depend on what the pressing concerns and immediate challenges are. It could possibly be a need to reduce energy consumption for instance. If this is the driving criterion, the indicator/s of relevance here would be the energy consumed in water treatment per unit volume water supplied, energy consumed in pumping per unit volume water supplied, energy consumed in wastewater treatment per unit volume wastewater collected and treated, and finally thereby, the total system-wide energy consumption per unit volume water supplied.

It is worthwhile to restate the caution advised by Yepes and Dianderas [100] against the use of too many or too few sustainability indicators. The use of too many of them is likely to dilute the power of all of them, while the use of too few may not adequately describe the utility's performance and progress in reaching its goals. If the selection is done wisely and benchmarks are set, the indicators play a key role in enabling utilities to monitor their performance over time and design appropriate course-corrective strategies. Converting indicators to indices or a single index may not really be meaningful if the main purpose is to measure progress and devise ways and means to improve further. Besides such a conversion would entail the choice of weighting /priority factors for the indicators of concern – this, it goes without saying, is contextual and there often may not be unanimity among a team of decision-makers as regards the same. A pro-environmentalist, for instance, is prone to canvass and argue for higher weighting factors for the environmental indicators, while an economist may beg to differ.

5.2. Research Question 2

The question is recalled here again:

> Over time, the challenges which utility managers encounter keep changing in form and degree of complexity. While ageing of assets is a prime concern, the environmental performance of urban water systems has increasingly come under the scanner in the recent past. Asset management in future entails taking the socio-cultural, environmental, economic, politico-legal and technological aspects into consideration.
>
> a) How does performance change over time, and how is it linked to the system's physical state, ageing of assets, changing operation and rehabilitation practices, and corresponding changes in the metabolism of resource inflows?

b) What characterises the present sustainability performance of Oslo's urban water system - as a case study – with respect to the major performance challenges and the reasons for the same?

c) How do these challenges affect/influence the social, economic and environmental sustainability of the system? How can they be overcome?

d) Why is it important to adopt a systems approach as far as sustainable asset management in Oslo's urban water system is concerned, and not look at the component sub-systems as 'islands of development'?

Answer to (a): The change in the functional/economic/environmental performance over time can be documented by resorting to a time series of indicators related to material and energy consumption, emissions and costs. The changes in selected indicators have been tabulated in *Table 5.1* and *Table 5.2*. The indicators of only the two extremes of a given time period have been listed. The intermediate values do not necessarily lie in between these two. In other words, there is not necessarily a trend of a consistent increase or decrease over time.

Both the water supplied and wastewater treated per-capita of the resident population decreased – the former by 15% and the latter by 9% over the eight-year period 2000-2007. The reasons could have ranged from measures undertaken by consumers to reduce water usage – installation of water-saving devices to a change in the demography of the population to a small reduction in the leakage rate courtesy rehabilitation of old water pipelines.

With both the water and wastewater pipeline networks being almost saturated, the annual environmental impact scores for both of them dropped over the 16-year period. It was the production and installation of a pipeline that accounted for the largest chunk of impacts among all the stages in its life-cycle (Venkatesh et al [121] and [122]). As installation activity has dwindled down, rehabilitation of old pipelines has now taken over as the dominant operation. The overall impacts decreased by 86% in the case of the water pipeline network and 96% in the case of the wastewater pipeline network. Specific energy and chemicals consumption in the WTPs increased by 25% and over 900% respectively, during the 10-year period 2000-2009. The reason, as pointed out in Venkatesh and Brattebø [123], was the decision to improve the level of treatment at the Oset treatment plant.

The specific energy consumption at the WWTPs rose by 0.17 kWh per m^3 in year-2007 with respect to year-2000. This was largely due to the rise in the availability of heat energy by the combustion of biogas. As mentioned earlier in *Chapter 4,* the 'consumption' in this case is not the same as 'actual heat demand'. The specific chemicals consumption dropped substantially over the same period – by 36%, owing to progressive optimisation of usage.

Performance indicator / aspect	Comments on changes over time
WTPs	
Per-capita water treated and supplied	Year-2000: 184.2 m^3 p.c. Year-2009: 167.7 m^3 p.c.
Energy consumption per unit volume	Year-2000: 0.212 kWh per m^3 Year-2007: 0.264 kWh per m^3
Energy and chemicals costs per unit volume	Year 2000: 1.25 year-2007- €¢/m^3 Year 2009: 4.5 year-2007- €¢/m^3
Chemicals consumption per unit volume	Year-2000: 10.6 grams per m^3 Year 2009: 97.4 grams per m^3
Specific environmental impact score– energy use	Year-2000: (1.41E-14) per m^3 Year-2009: (9.29E-14) per m^3 using Norwegian electricity mix
Specific environmental impact score– chemicals use	Year-2000: (4.22E-14) per m^3 Year 2009: (4.37E-13) per m^3 using Norwegian electricity mix
Water pipelines	
Total annual environmental impact score	Year 1991: (1.34E-5) Year 2006: (1.85E-6)
Water pumping	
Total energy consumed for pumping	Year-2000: 16.6 GWh Year-2009: 20.3 GWh
Entire upstream sub-system	
Specific energy consumption in the O & M phase	Year 2000: 0.39 kWh per m^3 Year 2006: 0.44 kWh per m^3
Operational expenses	Year 2004: 26.3 million year-2007-€ Year 2009: 28.7 million year-2007-€
Capital investments	Year-2004: 37.7 million year-2007-€ Year-2009: 19.9 million year-2007-€

Table 5.1: Changes in selected performance indicators in the water treatment and distribution sub-system in Oslo

Performance indicator / aspect	Comments on changes over time
WWTPs	
Per-capita volume of wastewater treated	Year-2000: 235 m^3 p.c. Year 2007: 198 m^3 p.c.
Specific energy consumption	Year-2000: 0.61 kWh per m^3 Year-2007: 0.78 kWh per m^3
Specific energy and chemicals costs	Year 2000: 4 year-2007- €¢ / m^3 Year 2007: 4.5 €¢ / m^3
Chemicals consumption per unit volume	Year-2000: 189 grams per m^3 Year-2007: 131 grams per m^3
Specific environmental impact -energy use	Year 2000: (8.85E-13) per m^3 Year 2007: (1.76E-12) per m^3 Using Norwegian electricity mix
Specific environmental impact- chemicals use	Year 2000: 7.94E-13 per m^3 Year 2007: 3.87E-13 per m^3 Using Norwegian electricity mix
Sewage pumping	
Total energy consumed for pumping	Year 2000: 6.22 GWh Year-2007: 6.67 GWh
Wastewater pipelines	
Total annual environmental impact score	Year-1991: (3.31E-5) Year- 2006: (1.12E-6)
Entire downstream sub-system	
Specific energy consumption in the O&M phase	Year 2000: 0.69 kWh per m^3 Year 2006: 0.81 kWh per m^3
Operational expenses	Year 2004: 42 million year-2007-€ Year 2009: 39.9 million year-2007-€
Capital investments	Year 2004: 17.4 million year-2007-€ Year 2009: 32.7 million year-2007-€

Table 5.2: Changes in performance in the wastewater transport and treatment sub-system in Oslo

The specific expenditure on energy and chemicals taken together in the WWTPs (in fixed year-2007 currency units) in year-2007 was 12.5% greater than that in year-2000. For the WTPs, this indicator rose by about 260% - from 1.25 (year-

2000) to 4.5 €-cents per m^3 (year-2009) (Venkatesh and Brattebø [127]). The reason for the latter, as referred to in the previous paragraph, was the improvement in treatment levels at the Oset WTP.

If the specific energy consumption in the operation and maintenance phase (O&M) is considered (Venkatesh and Brattebø [126]), the indicator increased from 0.69 to 0.81 kWh per m^3 of wastewater treated for the downstream sub-system, and from 0.39 to 0.44 kWh per m^3 of water supplied for the upstream sub-system, during the period 2000-2006.

The environmental impacts attributed to chemicals consumption in WTPs increased, concomitant with the rise in the quantities of chemicals demanded for treatment (over the period 2000-2009); while the corresponding indicator for WWTPs decreased by almost 50%, thanks to a significant reduction in chemicals consumption from 189 to 131 grams per m^3 of wastewater treated (over the period 2000-2007). The aggregated impact score attributed to energy consumption rose for both WTPs and WWTPs over the respective time periods, in tune with the rise in the specific energy consumption in the treatment plants (Venkatesh and Brattebø [123] and [124]). If the environmental impacts caused by the treated effluent from WWTPs are clubbed together with the impacts attributed to chemicals and energy consumption, it is seen (*Figure 4.16)* that the total specific impact score (per m^3 of wastewater treated) remained almost steady at around 8E-12. Eutrophication was the major impact category– dwarfing all the others. The upshot of this would be that the utility owner and operator of an urban water and wastewater system like Oslo's ought to always focus on and consider the emissions to the effluent sinks as the major environmental concern, even at a time when greenhouse gases and climate change tend to command greater attention.

As seen from *Table 5.2*, while the operational expenses dropped and the capital investments increased for the downstream, the converse was true for the upstream – during the period 2004-2009. A similar inverse relationship between capital investments and O&M expenses has been uncovered for the wastewater pipeline network, in Ugarelli et al [25]. If the entire system is taken into consideration, the ratio of annual capital investments to annual operational expenses rose from 0.6 to almost 0.9 from year-2005 to year-2009. While the upgrading of the Oset WTP dominated the capital investment flows in years 2006 (47.9%) and 2007 (41%), the investments in rehabilitating the wastewater transport and distribution sub-system accounted for almost half of the total capital flows into the system in years 2008 and 2009. It must be mentioned at this juncture that in Venkatesh et al [121], rehabilitation for the period 1991-2006 has been considered, owing to the timing of the said paper – however, the said paper and Ugarelli et al [25], in the forecast for the period 2008-2027, have predicted a growth in rehabilitation activity in the wastewater pipeline network.

Several other performance indicators (see *Appendix 10*) can be defined, measured and calculated, depending upon the purpose and specific control targets. *Table*

5.1 and *Table 5.2* list the values of some selected indicators from the journal papers referred to in *Chapter 4.*

Answers to (b) and (c): The water and wastewater pipeline networks in Oslo are almost saturated and the proportion of pipelines needing rehabilitation and repair will increase in the future. In such networks which will not expand perceptibly in the future, rehabilitation will take centre-stage. Decisions regarding the apportioning of funds between rehabilitation and maintenance (Ugarelli et al [25]) in order to ensure that the money is well and wisely spent to optimise the performance of the system, will be paramount to the utility in the years to come. As has been said in Baird [133], if the required level of funding to maintain and replace ageing infrastructures does not keep pace with the replacement timing and costs, the gap in the necessary level of investment increases at an alarming rate. The author of the said article terms this an 'enormous elephant standing in the path of sustainability.' While consumers are willing to pay more, as gathered from a recently-concluded personal interview (Venkatesh et al [131] with Kristiansen [16]), the utility however does not wish to sacrifice the need for economic sustainability and optimisation of expenses at the altar of the consumers' willingness to expend more for water supply and sanitation services. When it concerns rehabilitation, the decision to substitute polyurethane for epoxy resin as a rehabilitation material, is an environmentally-sound one (*Appendix 7).*

Coming to leakages, water of course is not 'destroyed'. It continues in the hydrologic cycle. But then chemicals, energy and money consumed to treat and distribute this non-consumed water, are wasted and are irrecoverable. Leaking sewer pipelines tend to contaminate water bodies (eutrophying them essentially), while leaking water pipelines are an indirect cause of excessive consumption of resources at the WTPs and pumping stations. Maintaining the pipeline networks with a view to minimizing leakages is certainly expensive. Further, if one is blessed with abundant water supply and fails to think about the consumption of material and energy resources, one would possibly just look upon pipeline maintenance as a cost burden which could well be deferred to the future. As pointed out in Maxwell [134], leakages from pipelines also lead to erosion and geotechnical instability. According to Kristiansen [16], the utility has not really done any kind of a cost-benefit analysis to determine how attractive a return on higher investments in rehabilitating pipelines would be, with regard to a reduction in expenses on chemicals and energy for treatment and pumping. The pipeline networks in Oslo, when it comes to leakages among other performance aspects, are ranked along with the former Soviet bloc countries in Eastern Europe. The progress towards sustainability entails the need to move up the ladder in this regard and aim to be ranked alongside the best that prevail in the continent (Venkatesh et al [131]).

Further, in Venkatesh et al [131], the Director of Oslo VAV, has informed the authors of the referred-to interview, that approximately 535 million € per year have been earmarked for the next five years (years 2011-2015), as investments into the water and sanitation system in Oslo. About 30 million € will be expended on constructing a tunnel in central Oslo in order to divert what is now an overflow discharge into the Oslo fjord to the Vestfjorden Avløpselskap (VEAS) WWTP to its west, to further reduce the discharge of nitrogen and phosphorus into the fjord. The population of the city is likely to grow by nearly 33% (200,000) in the next two decades. This will necessitate expanding the capacities of the two WWTPs – VEAS mentioned above, and the Bekkelaget WWTP (BEVAS). At present, these two WWTPs are already worked beyond rated capacity. So, both the future expected rise in population, and the continued need to reduce overflow into the fjord, are the triggers in this case. While the handling capacity will surely increase, the degree of nitrogen removal is likely to be increased by 2% - 3%. The utility also has on the anvil, a pilot project to separate urine from the wastewater streams and recovering nitrogen therefrom, in a small undeveloped part of Oslo which will have housing and inhabitation in the near future. Biogas generates electricity at VEAS, while BEVAS has started selling biogas to the public transportation sector (thus contributing to a reduction in the demand for and use of diesel fuel). The sludge from the WWTPs has been finding good use as fertilizer, for its nutrient content. The downstream of the system is thus already a model of sustainability. By the admission of Kristiansen [16], however, a lot more needs to be done, considering that sustainable development is a continuous process towards an elusive goal. In other words, sustainability challenges are never totally overcome.

If the wastewater transport and treatment sub-system is considered, data from Statistics Norway [116] show that the ratio of annual investments (different from the cumulative annual depreciation and interest payments on capital borrowed) to annual operation and maintenance expenditure (OPEX) was well below unity, for the period 2004-2008. On the other hand, for the water distribution and treatment sub-system, for years 2006 and 2007, the investments were greater than the OPEX, putting the said ratio at 1.93 and 1.42 respectively (Venkatesh and Brattebø [127]). This was evidently because of the upgrading of the Oset WTP which was called for by a need to improve the quality of the supply water – a social (health-related) or rather, a socio-political imperative. The upgrading also entailed the consumption of more chemicals and energy (Venkatesh and Brattebø [123]). Changes wrought in the water treatment system included the introduction of disinfection by ultraviolet radiation as the so-called ‘second-barrier’, replacement of chlorine gas by sodium hypochlorite, and aluminium sulphate by poly-aluminium chloride (Venkatesh and Brattebø [123]). As far as potable water supply is concerned, the question of overdoing does not arise, according to Kristiansen [16]. It is a basic need and people have the right to get water of the highest quality. Whenever there are health-related mishaps associated with contaminated water, or losses of marine life associated with discharge of untreated or poorly treated wastewater, the question of ‘how much’ is ‘too much’ may arise, but the basic motivation then is to take all precautions possible to avert similar happenings in the future. The outbreak of diarrhoea in another Norwegian city - Bergen - owing to *giardia lamblia* in the drinking water was enough to forewarn Oslo VAV of a possibility of the same in Oslo, if suitable measures were not taken. Sustainable development, as the Director of Oslo VAV says, entails adhering to the two ‘Rs’ - doing the ‘right thing’ at the ‘right time’. It has also been suggested by Baird [133] that the risk of water infrastructure failure can be reduced by better allocation of the investments of capital to replace only what needs to be replaced – the right pipe at the right time.

Answer to (d): The need for, and the challenges associated with systems-thinking (a holistic approach in other words) in the case of urban water and sanitation systems, can be perfectly illustrated with the aid of some examples. It will help to recall at this juncture, Robert White’s definition of industrial ecology from Ehrenfeld [26].

As and when measures are undertaken to reduce energy consumption in the system, a narrow-minded approach cannot be adopted. In other words, energy reduction is good enough, but not at the expense of other considerations. If energy-efficient systems need to be installed, an overhauling of the existing systems would be called for. Investments would then be needed; and the utility may have to approach the consumers for financial support. Thus, while environmentally, energy consumption reduction would be something highly desirable, there may be some social backlashes to handle. The fees would have to be hiked for instance (and there is an opportunity cost associated with the additional amount of money consumers pay for water); or investments in energy efficiency improvements in pumping and treatment may mean that there aren’t enough funds to rehabilitate leaking pipelines on time. There will certainly be a reduction in the OPEX owing to reduced energy consumption, but the question then would be the length of the payback period of the investments made.

If a good chunk of the investments are channelled into the WTPs to introduce additional process equipment and the operational costs are also increased by way of consumption of more chemicals and energy with the sole aim of improving water quality, but nothing is done to reduce the leakages in the water pipeline networks, the value (in terms of expenses and material/energy consumption per unit volume supplied) of the water lost, increases.

Trans-materialisation - as for instance replacing chlorine with sodium hypochlorite or alum with poly-aluminium chloride (Venkatesh and Brattebø [123]); or epoxy resin with polyurethane (Sægrov [111]) - may be driven by either economic or environmental or functional concerns. It is rarely so that any change can bring about improvements in economic, environmental and functional performance at the same time.

Often global warming and climate change are considered in general, to be the key issues to be addressed. Consequently, the other environmental impacts tend to get ignored. Acidification, abiotic depletion, eutrophication, stratospheric ozone depletion, photochemical ozone creation and toxicities of different types must not be overlooked in favour of global warming, when mitigation measures are devised. In the case of wastewater pipeline networks, for instance, the authors have presented a kind of a sequel to [121] in [132], wherein the said impacts have also been determined in addition to global warming.

Efforts made to improve energy efficiency and reduce environmental emissions at the plant level are sometimes offset by wasteful consumption on the part of consumers, who it must be remembered, are the direct demand drivers of the system. This makes sustainable consumption and sufficiency as indispensable to sustainable development of the system as a whole, as sustainable production and efficiency.

In the context of any modern urban water and sanitation utility, systems-thinking is often easier talked-about than accomplished. However, the indispensability of systems-thinking for sustainable development is very well understood and appreciated by utility managers these days. For instance, as Kristiansen [16] has said in Venkatesh et al [131], 'Oslo VAV has seven departments within itself, and it is very crucial, when we put on our thinking caps, to coordinate actions and decisions on a sustainable development framework.'

5.3. Research process and outcomes

Firstly, as mentioned several times before in this thesis, all the in-vogue industrial ecology tools have not been applied in this research process to the sustainability studies of the urban water and wastewater system in Oslo. Further, some of the tools employed, have not been applied to all the components of the system. Certain phases in the sub-system life-cycle have been kept out of the analyses for a myriad of reasons, which have been enumerated in the earlier chapters and hence are not being repeated in this section.

As mentioned in The Economist [136], 'Data are becoming the new raw material for any business – an economic input almost on par with capital.' This economic input is processed into metadata – 'information about information'; bits of relevant information in other words. This metadata, when applied (akin to the sale of products and services generated by employing capital and physical raw materials), enables decision-making, growth for businesses, and on a larger scale, development and progress for national economies. Industrial ecology studies are data-intensive and often handicapped by the lack of easy access to reliable and comprehensive data. The data needs of analysts are different from what the data-generators (or owners) are able or willing to supply. There are at-source uncertainties, which propagate as primary data are crunched to generate useful information, by the application of proxy factors, and other 'constants'.

However, one of the intended aims of this research has been to show that investing in better data management is essential and vital if assets are to be managed, maintained and sustained in keeping with the triple-bottom line approach (Elkington [4]). It is, so to say, to propound the dictum – *Give us the data and we shall show you the way forward.* Further, this research succeeds in positing the industrial ecology tools as useful aids in sustainability studies of urban water and wastewater systems. They contribute to a structured and well-defined approach to such studies.

Some interesting observations can be made at this juncture. Oslo accounted for 11.8% of Norway's resident population, 19% of its nitrogen inputs and 12.1% of its phosphorus inputs to WWTPs, in the year-2006. However, its shares in the nitrogen and phosphorus discharges into the hydrosphere – the causes of eutrophication which has emerged as the key environmental impact for the system as a whole – were 8% and 2.8% respectively (BEVAS [20], VEAS [22] and Venkatesh and Brattebø [129]). This clearly indicates a much higher degree of nitrogen and phosphorus removal from the wastewater at BEVAS and VEAS vis-à-vis most other WWTPs in Norway.

Sodium hypochlorite has replaced chlorine as a disinfectant at the WTPs in Oslo. However, as mentioned in *Chapter 2*, Travaglia [68] has challenged the widely-held notion that such a replacement ensures greater safety in handling, transport and use. Zoubolis et al [70] have contended that polyaluminium chloride (PAX) is a more efficient coagulant than alum, and results in the production of treated water with lower turbidity and lower residual aluminium content. This finding justifies the substitution of a significant proportion of alum by PAX in Oslo's WTPs.

Racoviceanu et al [65] has claimed that the electricity use for water treatment is, in general less than that for water distribution and wastewater treatment. In the case of Oslo, however, contrary to the statement above, the electricity consumption in the WTPs varied between 18.8 GWh and 21.74 GWh in the period 2000-2006. Water distribution (pumping) demanded between 15.51 GWh and 16.52 GWh during the same period. Thus, water pumping electricity demand was less than that used in the WTPs. In keeping with Racoviceanu et al's [65] statement, the electricity consumed in the WWTPs varied between 23.9 GWh and 30.21 GWh during the same period, and was thus greater than the consumption in the WTPs (Venkatesh and Brattebø [126]).

Biehl and Inman [72] have categorised the O&M in a typical water treatment plant in year-2008 thus – salaries (35%), energy (34%), chemicals (16%), other materials (13%) and maintenance (15%). For Oslo, in year-2007, energy and chemicals accounted for 6% and 48% respectively of the total O&M expenses, with salaries, maintenance and overheads accounting for 46%. In wastewater treatment in Greece, according to Tsagarakis et al [73], the expenses on chemicals and energy accounted for between 4% and 8%, and 40% respectively, of the total O&M expenses, while in a study of Scandinavian WWTPs by Balmer [74], the corresponding shares (on an average) were 10% and 25%. In contrast to the findings from Balmer [74] and Tsagarakis et al [73], chemicals accounted for a greater share of the O&M expenses in Oslo's WWTPs, than energy (Venkatesh and Brattebø [126]). This can be explained by the fact that biogas capture and reuse as a source of both electricity and heat in VEAS and as a source of heat in BEVAS, reduces the energy bill of the WWTPs. However, it must be mentioned, as has also been clarified earlier, that the cost of generating the electricity from the biogas (the investment and the O&M costs of the digester, turbine and generator, and the associated piping and instrumentation within the plant) has not been taken into consideration.

In Balmer [74], the WWTPs studied had a dependence on the external grid for electricity ranging from 6% to 100%. In BEVAS and VEAS taken together, the dependence varied between 75% and 85% (see *Appendix 1-b*). According to Clauson-Kass et al [75], electricity sourced from the grid by the Avedore WWTP in Denmark in year-1998 contributed most to the global warming. Whether this was true or not for Oslo's WWTPs would depend upon the choice of the electricity mix while performing the LCA. The Nordic mix could overestimate the global warming caused by electricity consumption, while the Norwegian mix might underestimate it.

The Belgian case study in Lassaux et al [82] has established that in WWTPs, the wastewater discharge into the final sink, dominates the environmental impact score calculated for wastewater treatment. This has been proved to be true for the Oslo case in Venkatesh and Brattebø [124] and depicted graphically in *Figure 4.16.*

Water is metered in several countries in the world. Nistor [96] has referred to the change in household behaviour with the adoption of water meters. Kristiansen [16] has mentioned in Venkatesh et al [131] that as far as the profligacy of water use in Oslo is concerned, metering of supply water may soon be entrenched into the system. Kristiansen [16] has also informed about a pilot project involving 20 households in Oslo, where water consumption is being metered.

5.4. Directions for further work

Overcoming the limitations of this research, and resolving the constraints encountered, constitute the scope of further work. Obtaining the missing data, and including the aspects and elements which have been neglected (on the basis of insignificance or irrelevance) in this research in order to arrive at not just more comprehensive but also more accurate results, will be essential.

It is necessary to drive home the fact that systems are simply integrations of sub-systems, interrelated to different degrees. Changes in a sub-system, or a set of changes in a few sub-systems, affect the ones in which changes are not initiated, to different extents. These effects may be direct or indirect, beneficial fallouts (resulting in win-win situations) or offsets (necessitating compromises). If the sub-systems within the system – as defined – are interrelated, the system as such has backward and forward linkages to (and spill-over effects on) other systems in the economy; this means that alterations within the system may have far-reaching (in scope, space and time) economy-wide implications.

As has been said often, monitoring is not just a necessary handmaiden of science (technology, business and economics). It is in fact the real thing. Performance, quality and condition monitoring and analyses often result in new discoveries and provide fresh insights, which would otherwise have never surfaced. Just as the objectives of economic advisers are often very different from those of the decision-makers in the government, academic researchers often end up being at variance with administrators and business managers. While academic research needs to factor in the practical considerations and on-the-ground dilemmas often encountered by the administration, the latter needs to admit the importance of the former for its vitality.

As Tollan [137] has observed, the Norwegian urban water and wastewater sector, in general, has a few salient concerns on its agenda – parasites in drinking water, leakages in pipelines, flooding due to pipeline bursts, wastewater sinks (the sea and the fjords), source control of environmental pollutants which enter the water stream, the impact of water use and wastewater discharge on fisheries and the role of the urban water and wastewater sector in climate change. Future strategies for sustainable development would thus be influenced by, and based on, these concerns. In the editorial of the magazine Tollan [137] is associated with, there is a strong emphasis on the importance of cost-benefit analyses of improvement options – one of the key directions of further work as a continuation of this research. In the Preface to Corcoran et al [138], the chairperson of the UN Secretary-General's Advisory Board on Water and Sanitation has stressed on the promotion of strategic financial planning in all countries - at the national level – to maximise efficiency to improve coverage in the water and sanitation sectors. Corcoran et al [138] has argued that in terms of public spending on health issues by the government, investing in improved wastewater management and supply of safe water provides particularly high returns (indirect gains over the medium-term), if the investments are backed by careful and comprehensive integrated water and wastewater planning and management at municipal and national levels. In Tollan [137], while pointing out that in the 2008-2010 period, interest in water policies grew conspicuously, the author has maintained that even though water is abundant in some areas (that is applicable also to the case study of this thesis), it is hardly surprising that water should eventually begin to take on more economic recognition and financial value.

5.4.1. Towards a more-holistic sustainability index

As mentioned earlier, and tabulated in *Appendix 10,* the data gathered and the results obtained in this research can be used to define and record some indicators under the economic, environmental, functional and social criteria. Once the indicators are grouped, it is essential to ponder over the importance or otherwise, of aggregation. Suitable weighting factors for indicators under every criterion and for the criteria themselves, reflecting the relative degrees of importance of the different aspects of sustainable development to the managing utility, would then lead to a more-holistic sustainability index for the urban water and wastewater system as a whole. The indicators are not absolute values but reflect the change in a particular year with respect to a base or reference year in the past. They are essentially quotients (referred to as 'Q' in *Figure 5.1*) obtained by dividing the values for the year of interest by the corresponding values for the reference or base year with which the comparison is being made. This is very apt and appropriate as sustainable development essentially aims at improvement and betterment over time. This improvement could be interpreted as either an increase or decrease in the value of an indicator – and this depends on what is being measured by the indicator.

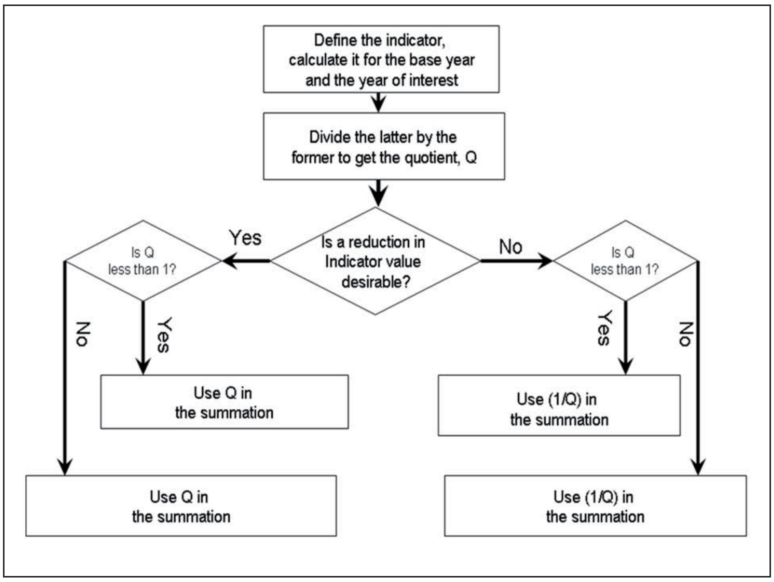

Figure 5.1: Flowchart to explain the choice between the quotient or its reciprocal for the averaging process

The weighted averaging could be done geometrically, arithmetically or as a mix of both methods. When the averaging is done, for every quotient Q taken into consideration, attention must be paid to whether an increase or decrease in the quotient is beneficial and desired. When a decrease in the value of Q is not desirable and Q is less than one, the reciprocal '1/Q' is used in the averaging process. Thus, by definition, a decrease in the value of the Holistic Sustainability Index below unity (which is, in effect the Index for the reference year), is deemed to be a movement towards sustainability and an increase above unity is considered to be unsustainable. *Figure 5.1* presents a flow chart which serves as a guide for the choice between Q and its reciprocal for each indicator quotient. The logic behind the application of the IF-THEN statements of *Figure 5.1* can be understood with simple examples. We start off with the premise, as referred to earlier, that a decrease in the Total Holistic Index to a value less than 1, is considered to be desirable, while an increase in it to a value greater than 1, hints at a movement away from sustainability. If the indicator for year-2010 is being compared to year-2006 for instance; and the quotient Q is obtained by dividing the former by the latter, we can consider four hypothetical possibilities to understand *Figure 5.1* better.

(a) Say for instance, the indicator is the *energy consumed per unit volume water treated.* Then, a decrease in this indicator is certainly desirable. So, if Q is less than 1, an improvement is indicated. Including this as-is in the summation will contribute to lowering the Index, which as mentioned in the premise signifies a move towards sustainability
(b) The indicator is still *energy consumed per unit volume water treated*, and the value of Q is greater than 1. This is certainly not desirable. Including Q now, as-is will contribute to an increase in the Holistic Sustainability Index. Again, in keeping with the premise, an increase indicates non-sustainability.
(c) If the indicator being considered is the *percentage of the resident population connected to the wastewater pipeline network,* an increase in the same over time is desirable. Or in other words, a reduction is not desirable. If Q in this case decreases, it is not acceptable. So, the reciprocal of Q – which then is greater than 1, is considered. Again, in keeping with the defined premise, this increases the Index, and that indicates non-sustainability.
(d) Considering the same indicator as in (c), if Q increases – in other words, the percentage of population connected to the pipeline network increases in year-2010 over that in year-2006, it is an improvement. We would then still take the reciprocal (as this would be less than unity), and in keeping with the premise, a value less than unity indicates a movement towards greater sustainability in year-2010 with respect to year-2006.

In the four equations below, the subscripts -'s', 'e', 'ec' and 'f - stand for social, environmental, economic and functional respectively; WGM stands for the weighted geometric mean, and WAM for the weighted arithmetic mean; 'a', 'b', 'c' and 'd' are the weighting factors respectively for the social, economic, environmental and functional sustainability criteria, such that the sum of the four factors is 1. '$SI_{(total)}$' stands for the holistic sustainability index, and the word following 'total' within the subscripted parentheses indicates the approach adopted to calculate the index. The 'hybrid1' approach calculates the weighted geometric mean of the weighted arithmetic means, while the 'hybrid2' approach does the converse.

$$SI_{total,aritmetic} = WAM_S \cdot a + WAM_{EC} \cdot b + WAM_E \cdot c + WAM_F \cdot d \quad \text{(Eq 5.1)}$$

$$SI_{total,geometric} = (WGM_S)^a \cdot (WGM_{EC})^b \cdot (WGM_E)^c \cdot (WGM_F)^d \quad \text{(Eq 5.2)}$$

$$SI_{total,hybrid1} = (WAM_S)^a \cdot (WAM_{EC})^b \cdot (WAM_E)^c \cdot (WAM_F)^d \quad \text{(Eq 5.3)}$$

$$SI_{total,hybrid2} = WGM_S \cdot a + WGM_{EC} \cdot b + WGM_E \cdot c + WGM_F \cdot d \quad \text{(Eq 5.4)}$$

Appendix 11 presents a hypothetical example, explaining the differences between the Holistic Sustainability Indices calculated by the four approaches. Quantifying sustainability by resorting to the use of indicators may lead to losses of important qualitative information which would have enhanced system understanding (Binder [30]). With an increasing degree of aggregation, information relevant to practical decision-making gets obfuscated, and the purpose of aggregation itself stands defeated. But, a given piece of information or processed data is useful in different forms to different entities for different purposes. Aggregation, it can be said, is meaningful and purposeful if it does not supplant the need for disaggregation, as and when required. This is quite akin to the aggregated environmental impact score in LCA studies where the normalisation and weighting factors introduce a lot of subjectivity to the final result.

Current practices of benchmarking and target-setting identify and measure selected indicators against preset benchmarks or targets – which can of course be continuously changed for progressive improvement. Aggregating the indicators by weighting and prioritising is generally not considered to be practical. According to Ai [105], the main goal of the Public Utilities Board of Singapore is to ensure a sustainable and diversified water supply to all the citizens of the country. The spokesperson named believes that it is not straightforward to quantify the weightings of the different indices (social, economic, environmental and functional), though the citizens of the country are constantly encouraged to conserve water, keep Singapore's water catchments and waterways clean and build a closer relationship with water. Metrowater, the utility in Auckland, New Zealand (Duke [106]) adopts the 'four well-beings approach' which includes, economic, environmental, social and cultural well-being, and while stating that the strategic objectives of the utility align with the protection of public health and safety, protection of the environment, enhancement of public services, provision for growth and optimisation of network integrity, informs that a criticality assessment method is being developed at the time of writing and that it would thereby be premature to recommend a schedule of weighting.

As Kristiansen [16] has said, 'sustainable development thinking' *per se,* is quite nascent as far as operations at the Oslo VAV are concerned. The utility has of course been pursuing several goals but has never attempted to integrate them or adopt a more holistic thinking. However, he has added that sustainable development is very much on the agenda of the utility in Oslo. There is also a keenness to develop robust and measurable metrics to measure progress towards sustainable development (Venkatesh et al [131]). The Holistic Sustainability Index – with carefully defined weighting factors for the indices and indicators - despite its demerits, can still be a useful metric for decision-making.

6. Conclusions

This research contributes to what Binder [30] refers to, as system knowledge. It shows the path ahead for action knowledge and effectiveness knowledge (both defined in Binder [30]) to evolve – supported by cooperative and collaborative ventures along with all the stakeholders in the system, to translate know-how and understanding into the much-needed change on a continuous basis. The main intent was to demonstrate the applicability and usefulness of the industrial ecology tools – as mentioned earlier in *Section 5.3* and the imperativeness of adopting a systems approach to the analysis and management of the urban water and wastewater system in general, and the one in the city of Oslo in particular. As summarised in *Table 3.1,* different analytical tools were applied to different components of the system.

The fact that all things are somehow interlinked, and a decision taken to improve one aspect of a sub-system may have either a detrimental or a positive effect on other aspects of the same sub-system or other sub-systems, is appreciated in principle. Adopting this knowledge in practice is however far from straightforward. The first step is to embrace the systems-thinking approach of industrial ecology to facilitate a better understanding of the said inter-linkages. Albert Einstein's famous quote **-** *A little knowledge is a dangerous thing; so is a lot* - can be recalled at this juncture (Clark [135]). Not knowing enough about the system and the interconnectedness of its elements leaves one clueless about the proper course of action for the future. However, knowing too much also leaves one befuddled and incapable of translating the knowledge into concrete action. The next section is a very brief summary of what has been discussed in the previous five chapters.

6.1. Summing up

This research was based on the water and wastewater system in the Norwegian capital city of Oslo as a case to test the usefulness of industrial ecology tools in sustainability studies. Oslo's water and wastewater system (surface-water based), managed by the Oslo *Vann og Avløpsetaten* (Oslo VAV) has been described diagrammatically in *Figure 1.2.* The city is serviced by 3 water treatment plants (WTPs) and 2 wastewater treatment plants (WWTPs). The system was broken down into its component parts – water pipelines, wastewater pipelines (sewage, stormwater and combined flow), water and sewage pumps, WTPs and WWTPs. While the primary raw data were sourced from Kristiansen [16], Brenden and Berger [17], Reksten [18], Toftdahl [19], BEVAS [20], Aasebø [21], VEAS [22] and Selseth [23], Statistics Norway [24] and personal communication with several experts in fields related to this research (referred to in *Chapter 2*) helped to fill in numerous data gaps. Literatures reviewed prior to and during the research have been touched upon in *Chapter 2.* Personal meetings and interviews with Kristiansen [16] and Reksten [18] towards the end provided useful additional insights into the workings of the system.

Having assembled the data to work with, the next step was to apply the industrial ecology tools – Material Stock and Flow Analysis, Energy Analysis, Embodied Energy Analysis, Environmental Life-Cycle Assessment and Life-Cycle Costing; and in addition to these, simple economic analysis and pipeline blockage analysis. The scope has been tabulated in *Table 3.1*. *Figure 1.4* is a schematic depiction of the metabolism in an urban water and wastewater system which the tools referred to, attempted to investigate, analyse and forecast.

From Venkatesh et al [121] and Venkatesh et al [122], it can be inferred that the water and wastewater pipeline networks in Oslo are almost saturated. In year-2006, there were 2.67 metres of water pipelines and 3.54 metres of wastewater pipelines per capita of the resident population. While installation of new pipelines has dwindled down to almost being negligible, rehabilitation has taken centre-stage. The environmental impacts, likewise, also decreased over the years 1991-2006 for both these pipeline networks, as has been depicted in *Figure 4.10*. For both the pipeline networks, global warming was the dominant impact during the study period. Ugarelli et al [25], while emphasizing the need for optimising the rehabilitation investments and operation and maintenance expenses, has posited a physical lifetime approach as a better rehabilitation strategy for wastewater pipelines (pipelines in general), than the in-vogue economic lifetime approach. In Ugarelli et al [125], sewage pipelines made of concrete, having a diameter between 150 to 180 mm, installed at a slope of between 0-15 ‰, and between 100 and 116 years old, have been identified as the ones most prone to blockages; and thereby the ones which deserve more attention with respect to rehabilitation. This paper thus was an advancement over Ugarelli et al [25], in that it advocated the need for condition monitoring and a thorough analysis of historical failures to ensure that the right pipe was rehabilitated at the right time.

An environmental life-cycle assessment of energy and chemicals consumption in WTPs (Venkatesh and Brattebø [123]) revealed that the impacts of chemicals increased dramatically after year-2007, courtesy a process upgrading at the Oset WTP. Except in years 2004 and 2007, the impacts attributable to chemicals consumption exceeded those due to energy consumption. Global warming was the key impact in both cases, though in the case of energy consumption, abiotic depletion and acidification were also quite significant.

If the overall environmental impacts from WWTPs are considered, it was the effluent wastewater which dominated, accounting for between 73% and 80% of the total impacts over the 2000-2007 time-period. While acidification dominated in the case of energy, and global warming accounted for the largest chunk of the aggregated score in the case of chemicals, eutrophication, not surprisingly, was the key impact caused by the effluent wastewater. It follows that eutrophication generally accounts for the lion's share of the total environmental impact score, and the scope for reducing the total environmental footprint by focusing merely on reducing greenhouse gas emissions, is thus very limited (Venkatesh and Brattebø [124]).

Moving over from the environmental aspect to the economic aspect, energy and chemicals for water and wastewater treatment, on an average, accounted for 10.8% of the total operational expenses in the water supply subsystem and 13.7% for the wastewater handling sub-system. There was a perceptible increase in this share from 5.2% in year-2004 to 14.9% in year-2009 for water; and 12.3 % to 14.2% for wastewater over the same time period. Chemicals cost more than energy for the WWTPs, while it was the other way round for the WTPs. The total real cost of energy and chemicals per m^3, in year-2007 currency, was between 4 € and 5.2 € for the WWTPs, and between 1 € and 4.5 € for the WTPs. The total (WTP + WWTP) per-capita real costs of energy and chemicals, expressed in year-2007 currency, rose from around 10 € in year-2000 to about 12.2 € in year-2007 (Venkatesh and Brattebø [127]).

An energy analysis of the entire system for flows, costs and impacts (Venkatesh and Brattebø [126]) revealed that the per-capita annual consumption of energy in the operational phase of the system varied between 220 and 260 kWh; the per-capita annual expenses on energy in inflation-adjusted year-2006-€ ranged between 6.5 € and 11 € and the per-capita annual GHG emissions ranged between 24 and 26 kilograms. The energy consumed on the upstream, per m^3 of water supplied was around 0.4 kWh on average, while the corresponding value for the downstream was 0.8 kWh per m^3 wastewater treated. The upstream GHG emissions ranged between 70 and 80 grams per m^3 of water supplied, about 22% greater on average than the corresponding specific GHG emissions on the downstream.

When the complex urban water and wastewater system in Oslo is broken down into its component parts, analyses using the IE tools become easier. When one would like to extend the scope and depth of the analyses however, the degree of complexity increases. The scope for simplification of the said performance analyses depends strongly on what Oslo VAV deems to be crucial for sustainability. Adoption of the tools thereafter, to study the stocks, flows and impacts over time, results in, *inter alia,* deliverables which can serve as useful metrics to measure performance. *Table 5.1* for instance, lists the values of some functional, economic and environmental indicators (and provides information at a glance, of the changes in performance over time). In the context of any modern urban water and sanitation utility like Oslo VAV, systems-thinking can be ingrained into the psyche of the administration, but it is not without its challenges. Kristiansen [16] has admitted in Venkatesh et al [131] that coordinating decisions and actions on a sustainable development framework is very crucial to the functioning of Oslo VAV in the future. He has expressed a keenness to develop robust and measurable metrics - like the deliverables of the

IE methods applied to the system - to measure progress towards sustainable development. That, in effect, could be within the scope of further work tailing this research.

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Appendices

Appendix 1: Pipe thicknesses

(a) **Thicknesses of concrete pipes (mm)** (Sægrov [111])

	Standards adopted in year						
Diameter (mm)	**1909**	**1919**	**1928**	**1936**	**1966**	**1970**	**1975 & later**
100	20	20	20	22	19	24	24
125	22	22	22	22	21	25	25
150	24	24	24	24	23	28	28
200	-	-	-	-	28	32	32
225	26	26	26	26	29	-	-
250	-	-	--	-	30	37	37
300	30	30	30	30	32	44	44
375	40	37	37	37	-	-	-
400	-	-	-	-	39	50	50
450	45	45	45	-	-	-	-
500	-	-	-	-	49	60	60
525	-	47	47	47	-	-	-
600	-	50	50	50	59	65	65
800	-	-	-	-	79	-	-
1000	-	-	-	-	80	-	-

(b) **Thickness of ferrous pipes (mm)** (Sægrov [111])

Diameter (mm)	**Grey cast iron (till 1953)**	**Grey cast iron (after 1953)**	**Ductile iron (used after 1965)**	**Mild steel**
75	9.7	8.6	6	3.1
100	9.9	9	6	3.5
125	10.4	9.5	6	-
150	10.9	10	6	4.5
175	11.4	-	-	-
200	11.9	11	6.3	5.8
225	12.5	-	-	-
250	13.2	12	6.8	6.2
300	14.5	13	7.2	7
400	16.5	15	8.1	7.5
500	18.5	17	9	7.5
600	20.3	19	9.9	7.5
800	23.4	21	11.7	9

(c) **Diameter/Thickness ratio of plastic pipes – 100 kPa rating** (Gjersø [107])

Material	(Diameter / Thickness) ratio
PVC	21
PE	11

(d) **Thickness of ferrous pipeline coatings and epoxy resin / polyurethane coatings** (Sægrov [111])

Ductile iron	Internal	After end of 1975	Cement mortar	Average of 6 mm Assumed
	External	Till end of 1975	Bitumen	Average of 5 mm of bitumen assumed
		After end of 1975	Zinc plus bitumen	175 g per square metre of zinc coating and 5 mm of bitumen assumed
		After end of 2000	Galvalume	Average thickness of 3.5 mils assumed for life of 50 years for suburban and industrial, colder, less-humid settings; Assumed 55% Al instead of the 5% Al Galfan
Mild steel	Internal	After end of 1975	Cement mortar	Average thickness of 6 mm
	External	All pipes	Bitumen	Average thickness of 5 mm
Grey cast iron	External	All pipes	Bitumen	Average thickness of 5 mm
Epoxy resin coating during rehabilitation: 7 mm				
Polyurethane coating during rehabilitation: 2-4 mm				

(e) **Estimates of backfill material masses during pipeline installations** (Sægrov [111])

As a rough estimate, an average depth of **3 metres** and an average width of **2 metres** can be assumed – this gives a cross-section of **6 square metres**. In case, there are two pipes in a trench – water and wastewater, it could go up to 9 square metres. A value of 7.5 square metres is assumed as an average value considering that there could be several cases of two pipelines in a trench. The specific gravity of gravel/stone is assumed to be 1.6. The porosity is neglected, resulting in an overestimate of the masses.

Appendix 2: Specific gravities of materials

Material	Specific gravity
Aluminium	2.50
Asbestos cement	1.20
Bitumen	1.02
Cement mortar	2.30
Concrete	2.30
Copper	9.00
Crushed stone/gravel	1.60
Diesel fuel	0.80
Ductile iron	7.10
Epoxy resin	1.08
Grey cast iron	7.10
Mild steel	7.85
Polyethylene	0.93
Polyurethane	1.05
Polyvinyl chloride	1.05
Zinc	7.14

Appendix 3: Diesel fuel consumption

	Small-size	Medium-size	Large-size
Installation	35 litres	40 litres	45 litres
Rehabilitation	1 litre	1.5 litre	2 litres

Values in per metre of pipeline installed or rehabilitated (Kristiansen [16])

Appendix 4: Data on water and wastewater treatment in Oslo

(a) WTPs (Aasebø [21])
Load on Oslo's WTPs

Year	Water supplied (million m^3)	Population of Oslo	Per capita water supplied (m^3 per capita p.a.)
2000	93.9	508,726	184.6
2001	93.3	512,589	182.0
2002	95.5	517,401	184.6
2003	92.8	521,866	177.8
2004	93.2	529,846	175.9
2005	94.1	538,411	174.8
2006	93.1	548,617	169.7
2007	95.1	560,849	169.6
2008	96.0	572,345	167.7
2009	98.0	584,292	167.7

Chemicals and energy consumed and sludge generated in Oslo's WTPs (Aasebø [21])

Material / Energy	**2000**	**2001**	**2002**	**2003**	**2004**	**2007**	**2008**	**2009**
	(all masses are in tonnes unless otherwise stated)							
Inflows								
$Al_2(SO_4)_3$	253	358	240	205	189	206	238	152
$Ca(OH)_2$	370	437	378	358	348	335	1,655	3,056
CO_2	351	336	320	313	326	286	1,632	3,104
Cl_2 gas	48.7	47.5	53.8	55.3	56.5	4.0	0	0
Microsand	0	0	0	0	0	0	128	282
PAX	0	0	0	0	0	0	1,200	2,640
Polymer	1.4	2.1	1.4	1.9	1.6	1.4	21.9	45.2
NaOCl	0	0	0	0	0	342	245	268
$^{\$}$Electricity (grid-GWh)	19.9	19.8	19.6	18.8	20.1	22.8	30.8	33.6
$^{\$}$Diesel fuel	0	0	0	0	152	180	186	191
Outflow								
Sludge solids* (million kg)	0.92	1.06	0.89	0.84	0.83	1.06	4.61	8.59

*Part of the sludge solids generated at the WWTPs
$^{\$}$Electricity consumed in years 2005 and 2006 was 23.5 GWh and 23.88 GWh respectively. Diesel fuel masses were 163 and 171 tonnes respectively.

(b) Load, energy and chemicals consumption in WWTPs in Oslo (Toftdahl [19], BEVAS [20] and VEAS [22])

Load on Oslo's WWTPs

Year	**Wastewater treated** (million m^3)	**Population of Oslo**	**Per capita wastewater treated** m^3 per capita p.a.)
2000	119.8	508,726	235.5
2001	110.3	512,589	215.3
2002	102.6	517,401	198.1
2003	105.6	521,866	202.4
2004	109.5	529,846	206.6
2005	111.4	538,411	206.9
2006	119.4	548,617	217.6
2007	111.5	560,849	198.7

Energy consumption in Oslo's WWTPs (Electricity in GWh)

Year	**Electricity Consumption**	**From grid**	**In-plant (from biogas-turbines)**
2000	30.97	23.87	7.10
2001	34.25	29.05	5.20
2002	37.49	30.12	7.37
2003	37.22	28.73	8.49
2004	37.82	29.73	8.09
2005	38.70	28.74	9.96
2006	38.99	30.21	8.78
2007	39.17	29.65	9.52

Energy consumption in WWTPs (Fuel for heating needs)

Year	**Heating oil** (GWh)	**Biogas utilised** (million m^3 at NTP)	**Biogas- derived heat** (GWh)
2000	1.70	8.12	40.23
2001	1.43	8.58	42.22
2002	1.16	10.56	41.23
2003	0.89	11.54	43.08
2004	1.39	11.87	45.99
2005	0.99	12.55	51.25
2006	1.21	12.66	49.54
2007	1.11	14.05	47.05

Masses of chemicals consumed and by-products generated in Oslo's WWTPs (masses in tonnes unless otherwise stated)

Material	2000	2001	2002	2003	2004	2005	2006	2007
Inflows								
Iron chloride	12,648	4,940	2,939	3,302	2,810	2,289	2,464	2,619
Iron sulphate	0	1,324	2,597	1,760	1,754	1,816	1,901	2,519
PAX	4,374	3,164	2,430	2,070	2,062	2,018	2,517	2,552
Methanol	2,084	2,121	2,179	2,297	2,048	2,225	2,304	2,398
Ethanol	0	11.8	12.0	19.7	19.2	19.8	1.5	0
Nitric acid	1,396	1,464	1,833	1,833	1,852	1,729	1,927	2,040
$Ca(OH)_2$	2,108	2,502	2,353	2,240	2,172	2,344	2,302	2,349
Polymers	97	65	80	85.1	87	83	75	88
Key outflows								
Sludge solids* (million kg)	5.9	5.2	6.4	7.4	7.7	5.3	5.4	5.0
NH_4NO_3	1,871	2,109	2,403	1,567	1,911	2,213	2,421	2,788
Grit & sand	**	**	**	**	2,299	2,322	2,066	1,955

*Sludge solids outflow includes the solids from WTPs also (between 10% and 21% of the total)
**Not known for BEVAS, hence total not depicted

Appendix 5: Embodied energy coefficients for pipeline materials (Ambrose et al [34] & Ambrose [103])

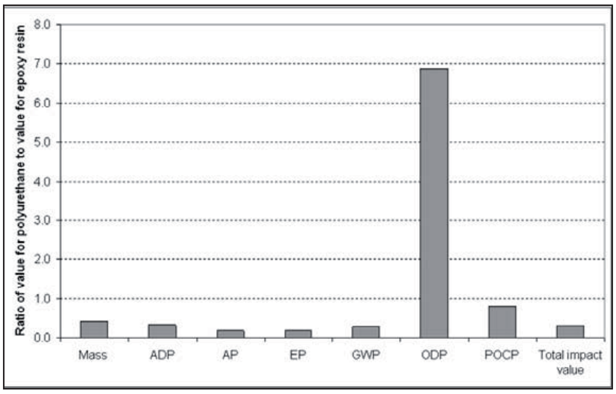
Pipelines made of…	Embodied energy coefficient (MJ/kg*)
Concrete	2
Grey cast iron	25
Ductile iron	38.2
Mild steel	96
PVC & Polyethylene (average)	85

*These are values for embodied energy in the material production and pipe fabrication phases

Appendix 6: Nordic electricity mix and the non-renewable fossil energy component in the same (Swiss Centre for Life-cycle Inventories [39])

Energy carrier	Shares of energy carriers as percentage of total				
	Norwegian	**Swedish**	**Danish**	**Finnish**	**Nordic mix**
Hard coal	0.03	1.29	51.20	20.35	**10.68**
Oil	0.01	1.21	11.20	0.80	**1.87**
Natural gas	0.32	0.28	23.50	14.80	**6.13**
Industrial gas	0.05	0.73	0.00	1.27	**0.61**
Hydropower	99.06	55.03	0.09	16.70	56.90
Wind	0.27	0.31	12.40	0.13	1.55
Wood & cogeneration	0.26	2.50	1.61	10.57	3.73
Nuclear	0.00	38.60	0.00	25.86	21.91
Lignite & peat	0.00	0.05	0.00	10.99	**2.57**
				Total fossil fuel component	**21.86**

Appendix 7: Polyurethane versus epoxy resin as pipeline rehabilitation materials (Swiss Centre for Life-Cycle Inventories [39] – dataset for Polyurethane, rigid foam, at plant/RER S and Epoxy resin, liquid, at plant; and Sægrov [111])



Appendix 8: Energy consumption in water and sewage pumping (Brenden and Berger [17] and Reksten [18])

Year	**Water pumping** (GWh)	**Sewage pumping** (GWh)
2000	16.06	6.22
2001	15.95	5.73
2002	16.33	5.33
2003	15.87	5.57
2004	15.94	5.69
2005	15.51	5.79
2006	16.52	6.21
2007	15.19	5.80
2008	16.26	6.32
2009	20.28	6.67

Appendix 9: Consumer price indices (inflation rates) and exchange rates between EUR and NOK.

Year	2000	2001	2002	2003	2004
CPI (1998=100)	105.5	108.7	110.1	112.8	113.3

Year	2005	2006	2007	2008
CPI (1998=100)	115.1	117.7	118.6	123.1

(Statistics Norway [116])

Year	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009
EUR /NOK	8.1	8.0	8.4	8.0	7.5	8.1	8.1	8.7	8.5	8.4

(Norges Bank [113])

Appendix 10: Sustainability (performance) indicators classified on the basis of components of the system and criteria

Component	Criteria	Indicators	Units
Water source	Economic	Cost of bringing new water sources on-stream, per unit volume additional supply required	Euro per cubic metre
		Estimate of cost of rainwater harvesting per unit volume water harvested without dependence on the municipal water supply	Euro per cubic metre
	Environmental	Water extracted, treated and supplied as a percentage of annual precipitation	%
		Groundwater level	Metres below surface
Water treatment plants	Economic	Annual operation and maintenance expenses (all expenses other than depreciation) per unit volume wastewater treated	Euro per cubic metre
		Annual capital-related expenses (depreciation) per unit volume wastewater treated	Euro per cubic metre
		Ratio of total annual costs to income generated from fees and sale of recovered chemicals or by-products if any	No unit
		Percentage of total income generated by means other than fees from the consumers	%

Water treatment plants	Environmental	GHG emissions to the atmosphere (measured differently for different GHGs) from the operation phase of the plant per unit volume wastewater treated	Kilograms per cubic metre (for each contributor)
	Functional	Cost of energy consumed per unit volume water supplied	Euro per cubic metre
		Cost of chemicals consumed per unit volume water supplied	Euro per cubic metre
		Mass of chemicals consumed in total per unit wastewater treated	Kilograms per cubic metre
		Total energy consumed per unit volume water supplied	kJ (or kWh) per cubic metre
		Sludge generated per unit volume water supplied	Kilograms per cubic metre
		Land area per unit volume water treated and supplied	Square metres per cubic metre
Water and wastewater pipeline network	Economic	Annual investments in rehabilitation of pipelines	Euro per year
		Expenses on operation and maintenance per unit length of pipeline in the network	Euro per kilometre
	Environmental	Annual environmental impacts associated with installation, rehabilitation, operation and maintenance of pipelines	Normalised, weighted and aggregated scores
		Total masses of pipeline materials in water and wastewater pipeline networks taken together per capita serviced by the network	Kilograms per capita

Component	Criterion	Indicator	Unit
Water and wastewater pipeline network	Functional	Rehabilitation rate in water pipeline networks	%
		Rehabilitation rate in wastewater pipeline networks	%
		Leakage rate in water supply network	%
		Leakage rate in wastewater pipelines	%
		Flooding events per year	Number of events
Wastewater treatment plants	Economic	Annual operation and maintenance expenses (all expenses other than depreciation) per unit volume wastewater treated	Euro per cubic metre
		Annual capital-related expenses (depreciation) per unit volume wastewater treated	Euro per cubic metre
		Ratio of total annual costs to income generated from fees and sale of by-products	No unit
		Percentage of total income generated by means other than fees from the consumers	%
	Environmental	GHG emissions to the atmosphere from the operation phase of the plant per unit volume wastewater treated	Kilograms per cubic metre
		Nitrogen in influent which goes out with the treated effluent	%
		Phosphorus in influent which goes out with the treated effluent	%
		Concentrations of heavy metals in the effluent (the key ones determined separately)	Milligrams per cubic metre

Component	Criterion	Indicator	Unit
Water and sewage pumping stations	Functional	Energy consumed per unit volume fluid handled	kJ (or kWh) per cubic metre
	Economic	Operation and maintenance expenses per unit volume fluid handled	Euro per cubic metre
		Annual capital investments per unit volume fluid handled	Euro per cubic metre
	Environmental	GHG emissions per unit volume fluid handled, due to energy consumption in the operation phase	Kilograms of CO_2-eq per cubic metre
System-wide	Social	Percentage of average income paid as fees for water and sanitation services	%
		Percentage of population connected to water treatment plants	%
		Percentage of population connected to wastewater treatment plants	%
		Cases of water-borne diseases per 10,000 people serviced	Number of instances
		Percentage of population which has installed water saving devices of some kind or the other to reduce water consumption	%
		Water taken up by illegal water connections (so-called Non-Revenue Water) as a percentage of total water treated and supplied	%
		Per capita water consumption in households	Litres per capita per day or per year

Component	Criterion	Indicator	Unit
System-wide	Social	Percentage of individuals or households who are willing to pay more to better the services when asked for	%
		Percentage of individuals served by the water and sanitation system who are happy with the services	%

<u>Appendix 11:</u> Hypothetical example of calculation of Holistic Sustainability Indices using the four approaches defined in *Chapter 6*

Assume that reduction is desirable for all the indicators; and the policymakers decide that 9 indicators are sufficient. Equal-weighting –intra-criterion and inter-criterion – is adopted. Year 2010 is the year for which the Index is to be calculated and Year 2006 is the base year.

$\frac{S_{(1)2010}}{S_{(1)2006}}$	$\frac{S_{(2)2010}}{S_{(2)2006}}$	$\frac{EC_{(1)2010}}{EC_{(1)2006}}$	$\frac{EC_{(2)2010}}{EC_{(2)2006}}$	$\frac{E_{(1)2010}}{E_{(1)2006}}$	$\frac{E_{(2)2010}}{E_{(2)2006}}$	$\frac{E_{(3)2010}}{E_{(3)2006}}$	$\frac{F_{(1)2010}}{F_{(1)2006}}$	$\frac{F_{(2)2010}}{F_{(2)2006}}$
1.1	**0.99**	**0.97**	**1.2**	**0.96**	**0.93**	**1.4**	**1.3**	**0.9**
WAM_S = 1.045 WGM_S= 1.043		WAM_{EC} = 1.085 WGM_{EC}= 1.078		WAM_E = 1.096 WGM_E= 1.076			WAM_F = 1.1 WGM_F= 1.081	
SI (total, A) = 1.0815 **SI (total, G) = 1.0693** **SI (total, A/G) = 1.0813** **SI (total, G/A) = 1.0695**								

A = arithmetic averaging; G = geometric averaging: A/G = arithmetic followed by geometric; G/A = geometric followed by arithmetic

The geometric averaging method indicates a 6.9% ‘worsening’ of the sustainability situation in 2010 as compared to 2006, while the arithmetic averaging one indicates a slightly higher ‘worsening’. The hybrid methods fall in between the two extremes.

Appendix 12: Corrigenda in appended journal papers (Facts and technical details only)

1. **Paper no 1** in the set of papers appended: [25] in List of References

Page 2281, Column 2: 'There are 64 wastewater pumping stations in the city…'
Page 2282, Table 2: The column headings – 3rd, 5th and 7th, refer to the Percentage of Stock of the respective material cohort
Page 2287, Column 1, Equation (2): It should be A_{rt} and not A_{rehab}
Page 2288, Column 2: 'The total investment will increase from about 198 MNOK in 2008 to over 204 MNOK in 2027.'

2. **Paper no 2** in the set of papers appended: [121] in List of References

Page 547, Column 2, Discussion: 'The network is now 37 years old'
Page 544, Figure 4: The physical lifetime approach is below and the economic lifetime approach is above.

3. **Paper no 3** in the set of papers appended: [123] in List of References

Introduction and Literature Review, first paragraph: Alunsjøen WTP used to be operational during the time period considered for the analysis. Hence, the reference to the same, along with Oset, Langlia and Skullerud. However, the tense used should have been the Past Tense for Alunsjøen. *Table 1* presents a snapshot of year-2009, when the Alunsjøen WTP, as per Per Kristiansen [16], has ceased functioning. Any reference to energy and chemicals consumption in Alunsjøen in 2009, or for that matter, erroneous references to 'four water treatment plants at the time of writing', may please be overlooked.
Page 16 in the enclosed PDF: Last line of the first paragraph: '…..the specific aggregated environmental impact score due to chemicals consumption increasing to 4.37E-13.'

4. **Paper no 5** in the set of papers appended: [125] in List of References

Table 2: Blockage rates are in blockages per kilometre per year; Blockage factor is a ratio of blockage rates without units.
Figure 3: Blockage factor on the Y-axis is a ratio of blockage rates and so, without units
Page 6, Column 1: Blockage factor is a ratio of blockage factors without units.

5. **Paper no 6** in the set of papers appended: [126] in List of References

Page 6, Column 2, Line 31: 'While the upstream depicts a linear correlation with time with an R^2 of 0.85, the costs for the downstream also have a decent conformity to the linear equation…..'

6. **Paper no 8** in the set of papers appended: [130] in List of References

Figure 3 and Figure 4: The captions have to be swapped.

Paper no. 1: ([25] in List of references)
R. Ugarelli, G. Venkatesh, H. Brattebø and S. Sægrov. Importance of investment decisions and rehabilitation approaches in an ageing wastewater pipeline network. A case study of Oslo (Norway).
Water Science and Technology. 58(12):2279-2293, 2008

Is not included due to copyright

Paper no. 2: ([121] in List of references)
G. Venkatesh, J. Hammervold and H. Brattebø. Combined MFA-LCA of wastewater pipeline networks – Case study of Oslo (Norway). ***Journal of Industrial Ecology***. 13(4):532-550, 2009.

Is not included due to copyright

Paper no. 3: ([123] in List of references)
G. Venkatesh and H. Brattebø. Environmental impact analysis of chemicals and energy consumption in water treatment plants: Case study of Oslo, Norway. ***Water Science and Technology - Water Supply.*** Accepted for publication on the 31st of July 2010.

Is not included due to copyright

Paper no. 4: ([124] in List of references)
G. Venkatesh and H. Brattebø. Environmental impact analysis of chemicals and energy consumption in wastewater treatment plants: Case study of Oslo, Norway.
Water Science and Technology – Water Supply
Accepted for publication on 31st July 2010.

Is not included due to copyright

Paper no. 5: ([125] in List of references)
R. Ugarelli, G. Venkatesh, H. Brattebø and S. Sægrov.
Historical analysis of blockages in wastewater pipelines in Oslo and diagnosis of causative pipeline characteristics.
***Urban Water*.** 7(6):335-343, 2010.

Is not included due to copyright

Paper no. 6: ([126] in List of references)
G. Venkatesh and H. Brattebø. Energy consumption, costs and environmental impacts for urban water cycle services: Case study of Oslo (Norway). *Energy*. 36(2): 792-800, 2011.
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Paper no. 7: [127] in List of references)
G. Venkatesh and H. Brattebø. Analysis of chemicals and energy consumption in water and wastewater treatment, as cost components: Case study of Oslo, Norway. Under review with ***Urban Water**,* 2010.

RESEARCH ARTICLE

Analysis of chemicals and energy consumption in water and wastewater treatment, as cost components: Case study of Oslo, Norway

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ABSTRACT

Adopting a systems-approach to an urban water and wastewater system, while applying a triple bottom line strategy to management, entails a careful analysis of all the sub-systems and components thereof with a view to improving service levels, optimising expenditure, augmenting investments, and also reducing the life-cycle environmental impacts associated with setting up, maintaining and operating the system. The scope for optimising expenses is system-wide, though it varies from one sub-system to another, depending on inherent lock-ins and external factors beyond the direct control of the water and wastewater utility. Optimising the consumption of energy and chemicals and improving the cost-efficiency thereof, is always on the agenda of WTPs and WWTPs. This paper analyses the consumption of, and the expenditure on chemicals and energy at Oslo's WTPs and WWTPs over time. Energy and chemicals for water and wastewater treatment, on an average account for 10.8% of the total operational expenses in the water supply sub-system and 13.7% for the wastewater handling sub-system. There is a perceptible increase in this share from 5.2% in 2004 to 14.9% in 2009 for water and 12.3 % to 14.2% for wastewater. Chemicals cost more than energy for the WWTPs, while it was the other way round for the WTPs. The total real cost of energy and chemicals per cubic metre, in year-2007 currency, was between 4 and 5.2 Euro cents for the WTPs, and between 1 and 4.5 Euro cents for the WTPs. The total (WTP + WWTP) per-capita real costs of energy and chemicals, expressed in year-2007 currency, rose from around 10 Euros in year-2000 to about 12.2 Euros in year-2007.

KEYWORDS: Investments, operation and maintenance expenditure, energy, chemicals, water treatment plants, wastewater treatment plants

INTRODUCTION AND LITERATURE REVIEW

Optimising expenditure and improving cost-efficiency (productivity with respect to cost) progressively are on the agenda of all businesses, households and governments. Urban water and wastewater utilities function primarily on the strength of the fees collected from the consumers whom they provide with the water supply and sanitation services. They rely also, for that matter, on loans when sizable investments are called for in order to enhance capacity, introduce state-of-the-art technology, retrofit existing equipment and machinery or automate operations. Utilities in some parts of the world (developing countries) where water supply and sanitation services are extremely cheap, have been expending more than what they have been raking in as income, and struggling to maintain their service levels (Zerah, 2007). The economic aspect is more crucial in developing countries as compared to the developed world, though of course, even in the developed world, optimising expenditure is certainly on top of the agenda of utilities. At present, the only water costs passed on to consumers concern transport or treatment. The scarcity of water is seldom reflected in its price. The revenue generated is often seldom enough to maintain or replace even existing infrastructure. (The Economist, 2010). Grau (1996) for instance, stressed on ‘affordability’ and ‘appropriateness’ when it comes to developing urban wastewater treatment systems. Van de Meene, et al (2009) pointed out that despite significant financial investment in urban water reform, the reforms have not been very successful, owing to numerous institutional barriers and lock-ins. However, in the article referred to earlier – The Economist, 2010 – it has been pointed out that man has been applying far more money to issues far less important than water; and that investing more cash and thought in the better use of the world’s most valuable commodity is surely worthwhile now. Harremoes (1998) observed that though water can be treated to ever greater degrees of purity at increasing cost, there is always a detectable residue which will be left behind. A cost-benefit analysis in this regard – from the water consumers’ point of view - would call for the estimation of the monetary benefits availed of by consumers by way of reduced expenses on healthcare and reduction in absenteeism from work, when water is treated to increasingly better degrees of purity. In Corcoran, et al

(2010), the authors argued that in terms of public spending on health issues by the government, investing in improved wastewater management and supply of safe water provides particularly high returns (indirect gains over the medium-term), if the investments are backed by careful and comprehensive integrated water and wastewater planning and management at municipal and national levels. Wastewater management, the referred-to publication says, has numerous environmental benefits, valuation of which is necessary to justify suitable investment policies and financing mechanisms. Analysts at Booz Allen Hamilton estimated in 2007 that US$ 22.6 trillion (in 2007 currency) would be required to upgrade the obsolescent water infrastructure in the world in order to meet the expanding demand between 2005 and 2030. Canada and the US would account for over 25% of this. (The Economist, 2010). Jackson (2009) pointed out that the ‘green stimulus’ element in the 2008 pre-Budget Report in the UK, GBP 25 million were allocated for flood defence and water infrastructure.

The prevalent scenario in most parts of the world today is that drinking water is cheap and therefore not given the same level of importance as say electricity which is metered and charged on a per-unit-basis. Costs play a significant role in shaping decisions. Nistor (2008) wrote about the situation in Moldova, that household consumption behaviour has been significantly affected by a rise in prices accompanied by the adoption of water meters. Maxwell (2010) while noting that an American family gets all the piped water it needs in a year for about USD 350, while buying the same amount of water from a vendor in the slums of Guatemala costs USD 1700, pointed out that often the poor pay a lot for their water needs (as a fraction of their incomes), while the rich pay next to nothing. In Oslo however, the utilities have the right to charge the consumers more whenever a need for additional investments in improving the service levels, arises. A comprehensive willingness-to-pay survey was conducted in Oslo a few years ago, and as told to the authors by Per Kristiansen, Director, Oslo VAV (the water and wastewater utility in Oslo) in a recent meeting (June 2010) all the Norwegians spoken to were and would still be happy to pay more if the service levels would improve as a consequence of the additional payments.

Efficiency improvements in the consumption of energy and chemicals, and service level augmentation would call for capital investments in incorporating state-of-the-art equipments into the setup. If the barriers identified by van de Meene, et al (2009), and referred to in the earlier paragraph, are removed, such investments should have the potential to progressively reduce operational expenses, and pay for themselves. Ugarelli, et al (2008) showed that this is possible for wastewater pipeline networks in Oslo, if the rehabilitation strategies are changed from the prevalent economic lifetime approach to a physical lifetime approach. Investments in the Oslo water and wastewater sector, as far as treatment is concerned, have primarily been driven by a need to upgrade the level of water treatment at the largest treatment plant in the city, and to enhance nitrogen and phosphorus removal from the wastewater in order to combat eutrophication of the fjord waters. Tsagarakis et al (2003*)* while discussing the application of cost criteria to select the right combinations of methods of wastewater treatment, found that the expenses on chemicals in wastewater treatment accounted for between 4% and 8% of the total operation and maintenance expenses on wastewater treatment in Greece, vis-à-vis 36% for energy. In a study of Scandinavian wastewater treatment plants, Balmer (2000) deduced that the costs of chemicals accounted for 10% of the total O & M expenditure and energy for 25%. In Balmer (2000), for the five WWTPs considered, the percentage of consumed electricity generated in-plant ranged from nothing to 94 per cent. Heinonen-Tanski et al (2000) while estimating costs for tertiary treatment of wastewater by rapid sand filtration with coagulants and ultraviolet disinfection, concluded that energy costs would account for 26 per cent of the operational expenses. In the present paper, the authors have determined these percentages for Oslo.

Keller et al (2003) while empasizing the need for harnessing the renewable energy potential of wastewater pollutants by resorting to anerobic processes to generate biogas, pointed out that the economics of energy recovery from sludge are governed by several constantly-changing factors, chief among them being the cost of generation in-plant and the cost of purchasing energy from the external market. In this regard, Sahely et al (2007) observed that in general, energy recovery from wastewater solids is not viewed as cost-effective, when electricity can be purchased relatively cheaply from the grid, or when fuel for heat energy is available at subsidised prices in the market. The electricity prices in Norway keep fluctuating and if there is a general trend towards a rise in prices, electricity generation from biogas at WWTPs would indeed become economically more attractive. Cornel et al (2009), while stressing on the feasibility of recovering phosphorus from wastewater plant sludge in Germany, put the additional costs for phosphorus recovery at 2 – 6 Euros per capita per year. This may be relevant when the sludge would otherwise be incinerated or land-filled. However, most of the sludge from the Oslo WWTPs is already being used for agricultural and landscaping purposes, for its fertiliser value. Chen, et al (2009) performed a cost-benefit analysis to determine the economic feasibility of decentralising wastewater treatment and reusing the treated effluent. Rebitzer et al (2003) performed an LCA-based life-cycle costing analysis to compare three different scenarios of wastewater treatment, and *inter alia*, has concluded that any activity carried out to improve the flocculation process and reduce the water-content of the sludge is highly beneficial to the economic bottom-line, as that would bring about savings in sludge transportation costs. Corcoran, et al (2010) quoting from a paper under review provided shadow prices reflecting the benefits associated with the removal of COD, BOD, nitrogen, phosphorus and suspended solids from wastewater for different effluent destinations - sea, wetlands, freshwater lake and reuse of wastewater, basing their analysis. Applying those shadow prices to Oslo's WWTPs to compare the benefits of treatment with the annual expenses incurred at the WWTPs, reveals that there is a very near break-even from years 2002-2006.

In this paper, an overview of the costs – expenditures and investments – in the water and wastewater system in Oslo is provided. Then, the scope is narrowed down to water and wastewater treatment plants and therein to the costs of chemicals and energy. A brief background of water and wastewater treatment in Oslo is provided, followed by a short outline of the methodology adopted in the paper. Results of the analysis of the consumption of, and the expenditure on chemicals and energy at Oslo's three water treatment plants (WTPs) and two wastewater treatment plants (WWTPs) over time, are subsequently presented. The specific expenses – real costs per unit volume fluid treated - are then determined and compared. Costs of energy and chemicals are then interpreted in terms of the total annual expenditure of the treatment plants (the sum total of operation and maintenance, salaries, interests on loans and rent payments, accumulated capital annual depreciation)

BACKGROUND

Overview of expenses, investments and income

The urban water and wastewater system in Oslo can be split up into four major anthropogenic constituent parts – water treatment plants, water distribution network consisting of water pipelines and water pumping stations (constituting the water supply sub-system upstream), wastewater transport network consisting of sewage, stormwater and combined flow pipelines and sewage pumping stations, and wastewater treatment plants (constituting the wastewater handling sub-system downstream). This system, on date, services about 590,000 people in the city of Oslo. Income generated from fees and sale of by-products enable the utility to finance its operation and maintenance expenses (*OPEX* in *Table 1*) and capital interest payments; while loans facilitate investments to upgrade the system. While interest payments account for one portion of the capital expenditure (*CAPEX* in *Table 1*), the annual depreciation of the capital investments account for the other.

All the monetary amounts in the following discussion are expressed in constant year-2007 Euros. It should be mentioned at this juncture that owing to the fluctuations in the exchange rates, the rates of change in the expenses, investments and income (when the monetary values are expressed in year-2007-Euros) are different from the actual rates of change in the corresponding year-2007-NOK values.

The total income gathered from fees rose from 95.1 million Euros in year-2004 (179.4 Euros per capita serviced) to 108.3 million Euros in year-2009 (185 Euros per capita serviced). The income generated for the water supply sub-system was about 40% of the total throughout this period. The average annual investment in both the sub-systems taken together was 47.8 million Euros. The investments into the water supply sub-system were greater than those into the wastewater handling sub-system from 2004 to 2008, ranging between 51% and 73% of the total. In year-2009 however, of the 52.6 million Euros invested, the wastewater handling sub-system accounted for 64%. Of the 287 million Euros invested during this six-year period, 71% was for treatment and 29% for the transport and distribution networks. In the water supply sub-system, the water treatment plants accounted for 45% of the investment, while for the wastewater handling sub-system, the wastewater treatment plants accounted for a little over 6%.

The average annual expenditure (CAPEX + OPEX) was 102.6 million Euros. The water supply sub-system accounted for between 38% and 48% of the total annual expenditure on the system. The OPEX was greater than the CAPEX in general for both sub-systems (except in the year 2007, when the capital investments in the water supply sub-system were slightly greater than the operational expenses). Wages in the water supply sub-system accounted for a greater share of both the annual sub-system expenses and the operational sub-system expenses, vis-à-vis those in the wastewater handling sub-system. For the entire system, wages accounted for between 29% and 39% of the total operational expenses and between 16% and 22% of the total annual expenses.

Energy and chemicals for water and wastewater treatment, on an average account for 10.8% of the total operational expenses in the water supply sub-system and 13.7% for the wastewater handling sub-system. There is a perceptible increase in this share from 5.2% in 2004 to 14.9% in 2009 for water and 12.3 % to 14.2% for wastewater (here, it is necessary to reiterate the influence of the fluctuating exchange rates). *Figure 1* depicts the partitioning of investments and expenses among the components of the water and wastewater sub-systems for year-2007.

Table 1: Income, investments and expenses in Oslo's water and wastewater system for the period 2004-2009 (in million year-2007-Euros)

	2004		2005		2006		2007		2008		2009	
	W	WW	W	WW	W	WW	W	WW	W	WW	W	WW
INCOME FROM CONSUMER FEES	**37.5**	**57.6**	**38.4**	**57.1**	**40.3**	**60.4**	**39.6**	**59.2**	**42.8**	**64.2**	**43.4**	**64.9**
INVESTMENTS	**37.7**	**17.4**	**25.4**	**13.3**	**30.8**	**11.5**	**32.3**	**14.5**	**26.0**	**25.3**	**19.9**	**32.7**
Treatment plants	*18.1*	*0.7*	*11.9*	*0.2*	*20.3*	*0.3*	*19.0*	*0.3*	*7.1*	*0.2*	*1.1*	*5.3*
Transport / distribution	*19.6*	*16.7*	*13.5*	*13.1*	*10.5*	*11.2*	*13.3*	*14.2*	*18.9*	*25.1*	*18.8*	*27.4*
ANNUAL EXPENSES	**41.5**	**66.7**	**45.6**	**54.0**	**43.4**	**53.4**	**44.0**	**57.4**	**49.0**	**56.9**	**49.3**	**54.5**
(of which) CAPEX	**22.7**	**15.2**	**19.6**	**16.9**	**21.3**	**21.9**	**22.4**	**24.4**	**23.9**	**23.8**	**20.6**	**14.6**
Capital interest payments	*5.2*	*7.7*	*6.8*	*5.9*	*10.7*	*10.5*	*12.5*	*9.0*	*13.0*	*8.7*	*9.1*	*6.4*
Capital depreciation	*10.0*	*15.0*	*12.8*	*11.0*	*10.6*	*11.4*	*9.9*	*15.4*	*10.9*	*15.1*	*11.5*	*8.2*
(of which) OPEX	**26.3**	**42.0**	**26.0**	**37.1**	**22.1**	**31.5**	**21.6**	**33.0**	**25.1**	**33.1**	**28.7**	**39.9**
Wages	*9.9*	*13.0*	*9.7*	*11.0*	*9.6*	*11.4*	*9.1*	*7.6*	*10.3*	*9.2*	*10.8*	*9.5*
Chemicals for treatment	*0.22*	*3.0*	*N.A.*	*2.7*	*N.A.*	*2.6*	*0.23*	*2.8*	*0.79*	*N.A.*	*1.42*	*N.A.*
Energy for treatment	*1.15*	*2.2*	*N.A.*	*2.6*	*N.A.*	*2*	*1.88*	*1.9*	*2.5*	*N.A.*	*2.87*	*N.A.*
Maintenance & overheads	*15.0*	*23.8*	*N.A*	*20.8*	*N.A*	*15.5*	*10.4*	*11.5*	*13.6*	*N.A.*	*15.0*	*N.A*

N.A. = Not Available; W = Water sub-system (includes treatment and distribution); WW=Wastewater sub-system (includes treatment and transport)

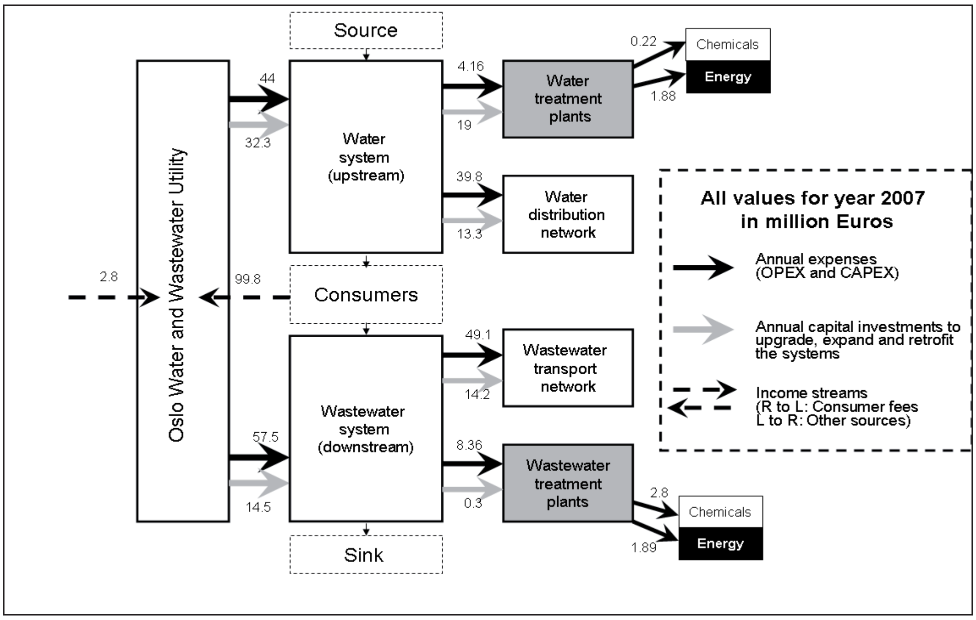

Figure 1: Money flows in the Oslo water and wastewater network for year 2007, in million Euros

Ageing pipeline networks – both water distribution and wastewater transport – call for significant investments (and thereby capital interest payments) to facilitate rehabilitation (Ugarelli et al, 2008), and as referred to earlier, these investments, if wisely directed, result in a reduction in operation and maintenance expenses incurred on the pipeline networks. The electrical energy (totally sourced from the grid) consumed for water distribution and wastewater transport is greater than that consumed during the treatment processes, and improving the efficiency of the pump-sets can lead to significant gains by way of reduced pumping energy consumption and thereby reduced expenditure on energy. Investments in water and wastewater treatment, as mentioned earlier, have been driven by requirements to enhance the degree of water treatment and the degree of nitrogen and phosphorus removal from the wastewater. The scope for cutting costs by wage reduction is minimal especially in a welfare state like Norway's. Regular and routine maintenance expenses can be trimmed down a bit (or at least the increase in the same can be arrested) by investing in upgrading and retrofitting components of the system. In other words, an increase in capital interest payments and annual depreciation can be partially offset by a drop in certain categories of operation and maintenance expenses. As far as interest rates on loans to service capital needs are concerned, the utilities cannot directly control the fluctuations in these, as there are determined, in general, by macroeconomic governmental fiscal policies.

Improving service levels entails, *inter alia*, treating water and wastewater to progressively-better degrees. Achieving this economically, entails increasing the efficiency of use of chemicals and energy. Optimising the use of these factors of treatment also yields environmental benefits. The focus of this paper is narrowed down to the expenses on chemicals and energy for water and wastewater treatment in Oslo.

Water and wastewater treatment in Oslo

Oslo's WTPs – three in number at the time of writing – supply, among themselves, close to 100 million m^3 of treated water, serving a population of about 590,000 inhabitants. The Oset WTP – the largest of them all - supplies 90% of the total, with Skullerud coming in a distant second, followed by Langlia. All the three plants do not treat the water in the same manner. While Langlia resorts to filtration and disinfection with sodium hypochlorite, Oset and Skullerud adopt microfiltration, chemical treatment, and disinfection with sodium hypochlorite and ultraviolet radiations (the so-called second line of hygienic barrier). However, before 2008, the methods of treatment at Oset and Langlia were nearly the same – except for the fact that Langlia used chlorine gas for disinfecting the water. After 2008 the consumption of chemicals has risen owing to the fact that 90% of the supply, which earlier was not subjected to chemical treatment, now consumes polyaluminium chloride, calcined lime, carbon dioxide, microsand and polymer, in addition to sodium hypochlorite.

There are two wastewater treatment plants (WWTPs) in the city - *Vestfjorden Avløpsselskap* (VEAS) and Bekkelaget *Vann og Avløpsselskap* (BEVAS). The VEAS treatment plant has pre-treatment with screening and aerated grit chambers, before coagulant (iron chloride / polyaluminium chloride) and polymer is added prior to primary sedimentation, followed by biological nitrification-denitrification and secondary sedimentation. The sludge treatment includes centrifuges followed by anaerobic sludge digestion that supplies biogas for electricity generation in-plant. Heat recovered from the exhaust gas is utilised for heating requirements within the plant by using sea water as a heat exchanger fluid. The sludge is conditioned and vacuum-dried, while the filtrate water from the drying unit is sent to an ammonia stripping unit, which is fed with nitric acid to remove the ammonia as ammonium nitrate. This is a useful by-product which finds use as a nitrogenous fertiliser. The dried, dewatered and digested sludge is a useful output from the facility, while the sand and the screened impurities from the upstream comprise another waste stream. At BEVAS, after screening and grit chambers, the flow is split into two. A portion of the flow (stormwater essentially) is only subject to physical and chemical treatment, while the sewage and combined flow undergo biological treatment as well. Iron chloride and polyaluminium chloride (PAX) are added to the storm-water flow which is then directed to the biofiltration units. The sewage passes through primary sedimentation, before entering biofiltration (aerobic and anaerobic treatment) and secondary sedimentation. The treated effluent from the secondary clarifier and the storm-water flow with the coagulants are then sent to sand filtration units. The biogas generated is used to supply the heat requirements of the treatment plant. The digested sludge is dewatered and conditioned before storage in silos, wherefrom it is trucked away to its end-use destinations. *Table 2* lists the volumes of water supplied and wastewater treated in Oslo, over time.

Table 2: Water supplied and wastewater treated in Oslo's WTPs and WWTPs

Year	**Water supplied (million m^3)**	**Wastewater treated (million m^3)**	**Population of Oslo (millions)**	**Per capita water supplied (m^3 per capita p.a.)**	**Per capita wastewater treated m^3 per capita p.a.)**
2000	93.9	119.79	0.508	184.6	235.5
2001	93.3	110.34	0.513	182.0	215.3
2002	95.5	102.52	0.517	184.6	198.1
2003	92.8	105.64	0.522	177.8	202.4
2004	93.2	109.45	0.530	175.9	206.6
2005	94.1	111.40	0.538	174.8	206.9
2006	93.1	119.38	0.549	169.7	217.6
2007	95.1	111.46	0.561	169.6	198.7
2008	96.0	----	0.572	167.7	----
2009	98.0	----	0.584	167.7	----

METHODOLOGY

Data for the two wastewater treatment plants were sourced from BEVAS (2007), VEAS (2001-2007) and personal communication with Toftdahl (2009). Aasebø (2009) provided data for the three water treatment plants in the city. The masses of chemicals and the amounts of energy consumed at the WTPs and WWTPs have been tabulated in *Appendix I.* The data for chemicals and energy consumption were not available for the WTPs for the years 2005 and 2006 at the time of carrying out this analysis. A fraction of the electricity consumed in the WWTPs (only at VEAS) was generated in-plant by combusting the biogas. Consumer price indices were obtained from Statistics Norway in order to represent the real costs, and the floating annual exchange rates (Norwegian Kroner to Euro) were obtained from Norges Bank, to enable a conversion from Norwegian currency to Euros. (Appendix II tabulates these values).

For the WWTPs, only the total expenses incurred on chemicals are known, while for the WTPs, unit prices of each of the chemicals is known (refer *Appendix I*) facilitating a break-up of the total expenses into component parts. The total expenses on electricity are known, while unit costs of heating oil and diesel are obtained from *Statistics Norway.* The unit cost of electricity generated from the biogas in-plant is assumed to be the average over the time period studied, of the annual tariffs paid for the electricity sourced from the grid. (Balmer (2000) has used a similar approach). Of course, there are studies like Eastern Research Group, Inc. & Energy and Environmental Analysis, Inc. (2007) which have calculated the life-cycle costs of different types of Combined Heat and Power systems in wastewater treatment plants in the USA. However, the authors refrain from using those values to calculate the cost of generating electricity from biogas at VEAS.

The recovered waste heat at VEAS is considered as a bonus, and the costs of the equipment utilised to capture and transport it can be considered to be a part of the electricity generation costs. For BEVAS, where there is no in-plant electricity generation, the cost of converting biogas into usable heat within the plant is ignored. As the focus is restricted to the actual expenses incurred by the plants, the avoided costs of purchase of natural gas (for heating applications) are not considered. Also, when one considers the fact that the available heat (waste heat recovered from the exhaust after electricity generation at VEAS and direct production by combustion at BEVAS) from the combustion of biogas is usually greater than the real demand for heat in the plants, it is difficult to estimate the exact amount of natural gas which would have been purchased in case the biogas-derived heat had not been available for use. (A quick calculation reveals that if all the biogas-derived heat at VEAS and BEVAS – tabulated in *Appendix I* - had been generated by combusting purchased natural gas instead, the additional nominal expenditure would have increased from 7.2 MNOK (849,000 Euros) in year-2000 to nearly 15 MNOK (1.73 million Euros; when the natural gas prices tabulated in *Appendix III* are considered) in year-2007. If these avoided costs are considered, then the net expenses on energy are depressed to a great extent). The small revenue which accrues to the treatment plants, thanks to the sale of by-products (sludge as fertiliser, ammonium nitrate etc.), is also not taken into consideration.

The nominal unit costs (expressed in Euros, by resorting to the exchange rates in *Appendix II*) for heating oil, diesel, natural gas and electricity are tabulated in *Appendix III*. The VEAS treatment plant handles wastewater from parts of Oslo, Asker and Bærum. Only the Oslo component of the influent wastewater into VEAS (which accounts for between 60 and 70% of the total) is considered and the total energy and chemical expenses are accordingly apportioned. The specific real costs in 2007-Euros are determined for chemicals and energy separately for WTPs and WWTPs. For WWTPs, this exercise is performed for VEAS and BEVAS separately for the sake of comparison, while for the WTPs, all three are clubbed together. The shares of the different chemicals in the total cost pie can be determined and commented upon for the WTPs. The expenditure on energy and chemicals is then expressed as a percentage of the total annual expenses, obtained for WWTPs (from VEAS (2001-2007) and BEVAS (2007)) and WTPs (Aasebø, 2009; Klemetsrud, 2010 and Skedsmo, 2010).

RESULTS AND DISCUSSIONS

The results are presented in terms of the specific real costs – costs in 2007-Euros adjusted for inflation, per unit volume water or wastewater treated. This will enable an easy comparison with similar results from other water and wastewater treatment plants.

Water treatment plants

The specific real costs for energy and chemicals taken together increased from around 1.25 Euro cents per cubic metre of water treated, to a little under 4.5 Euro cents per cubic metre in year-2009 (refer *Figure 2*). The doubling of the specific costs from year-2007 to year-2009 was due to upgrading of the treatment facility at Oset, which necessitated a greater consumption of energy and chemicals. Energy dominated the costs all along – annual expenses on energy were between 2 and 8 times greater than those on chemicals.

The shares of the different chemicals consumed at the WTPs, in the chemicals costs pie, changed over time. Sodium hypochlorite made a foray at the expense of chlorine in 2007, and PAX displaced aluminium sulphate to a great extent in 2008. As Figure 3 shows, chlorine gas by far accounted for the lion's share of the costs (between 30% and 40%) with calcium hydroxide following not very far behind from 2000 to 2004. In year-2007, sodium hypochlorite which almost phased chlorine gas out totally, accounted for over 40% of the chemicals costs, while PAX (over 30%) did so in years 2008 and 2009.

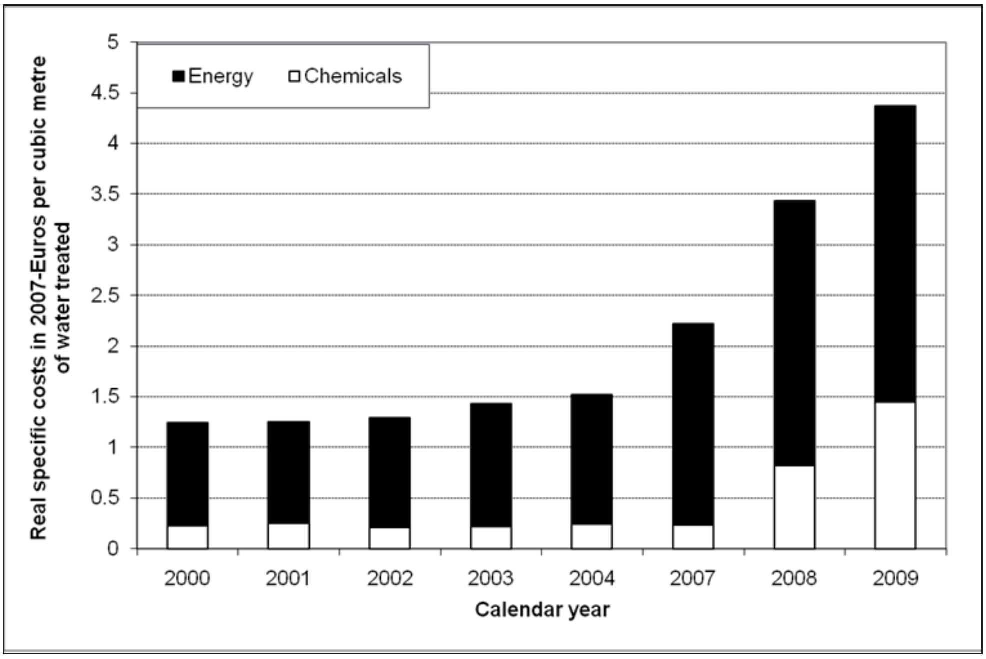

Figure 2: Specific real energy and chemicals costs in Oslo's WTPs

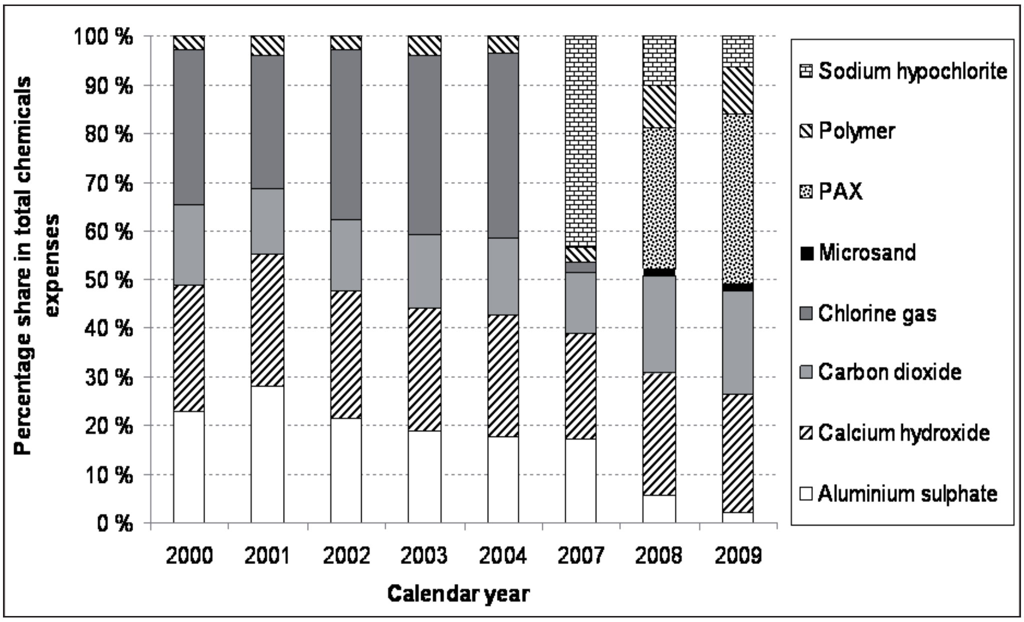

Figure 3: Shares of different chemicals consumed in Oslo's WTPs to the chemicals costs pie

Wastewater treatment plants

Overall, the expenses on energy and chemicals at the WWTPs, rose almost steadily from 4 Euro cents per cubic metre in 2000 to 5.2 Euro cents in 2005 before dropping to 4.7 Euro cents in 2007 (Figure 4). The initial rise was partly due to a decrease in the volumes of wastewater treated vis-à-vis year 2000, and probably due to a rise in the prices of some chemicals over the annual inflation rates.

The specific energy costs at BEVAS, as seen in *Figure 5* were less than those at VEAS from 2000 to 2003, and in 2005. The range during the 2000-2007 period was between 0.5 to 2.3 Euro cents per cubic metre of wastewater treated, at BEVAS, and between 1.75 and 2.8 Euro cents per cubic metre at VEAS. When both plants are taken together, the range was between 1.3 to 2.65 Euro cents per cubic metre. The drivers behind this increase were the rise in the price of electricity by about 50% over the period 2000-2007 (see *Appendix III*), followed by a 30% rise in electricity consumption (see *Appendix I*). The rise in electricity consumption was owing, *inter alia,* to the aeration needs for better nitrification-denitrification (eutrophication control being one of the key goals of wastewater treatment in Norway in general).

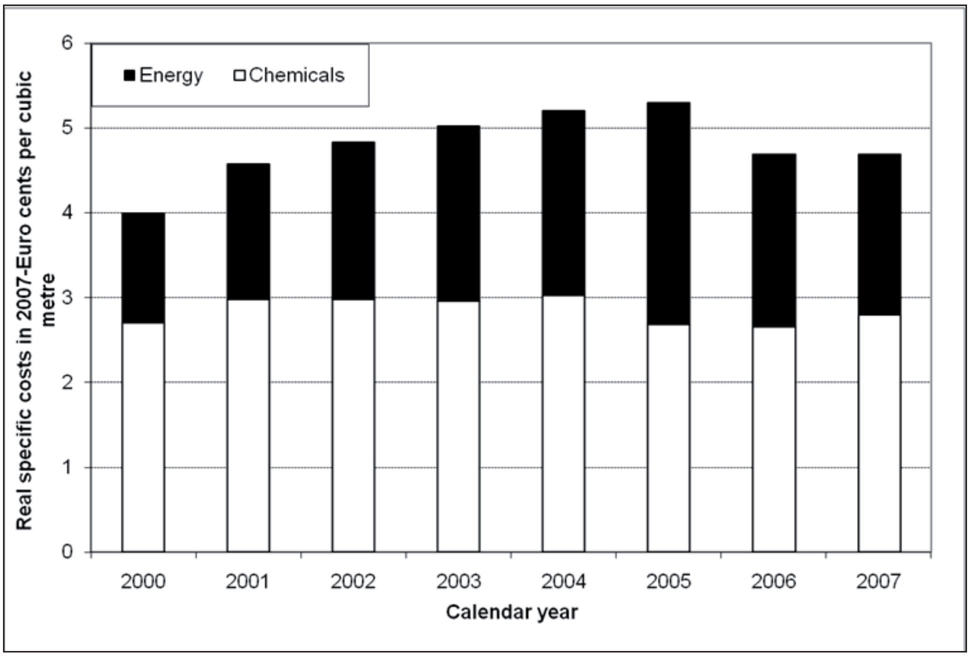

Figure 4: Specific real energy and chemicals costs in Oslo's WWTPs

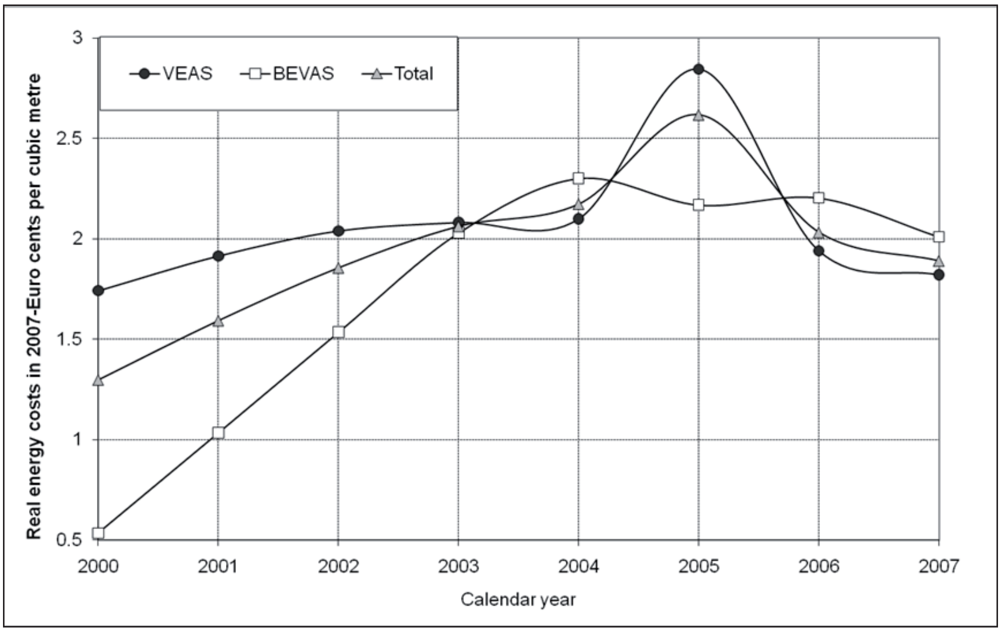

Figure 5: Specific real energy costs in Oslo's WWTPs

The specific chemicals costs at BEVAS, were higher than those at VEAS from year-2002 to year-2006. The range at VEAS was between 2.6 to 3.1

Euro cents per cubic metre, while for BEVAS, this ranged between 2.5 and 3.15 Euro cents per cubic metre (*Figure 6*). The sum of the specific costs of energy and chemicals at VEAS was greater than that at BEVAS in years 2001-2003, 2005 and 2007. The ratio of the former to the latter ranged between 0.98 and 1.47. Contrary to the WTPs, the chemicals costs were greater than the energy costs at the WWTPs. The reason behind this is not very difficult to pinpoint – the avoidance of purchase of natural gas for heating applications, thanks to in-plant biogas production

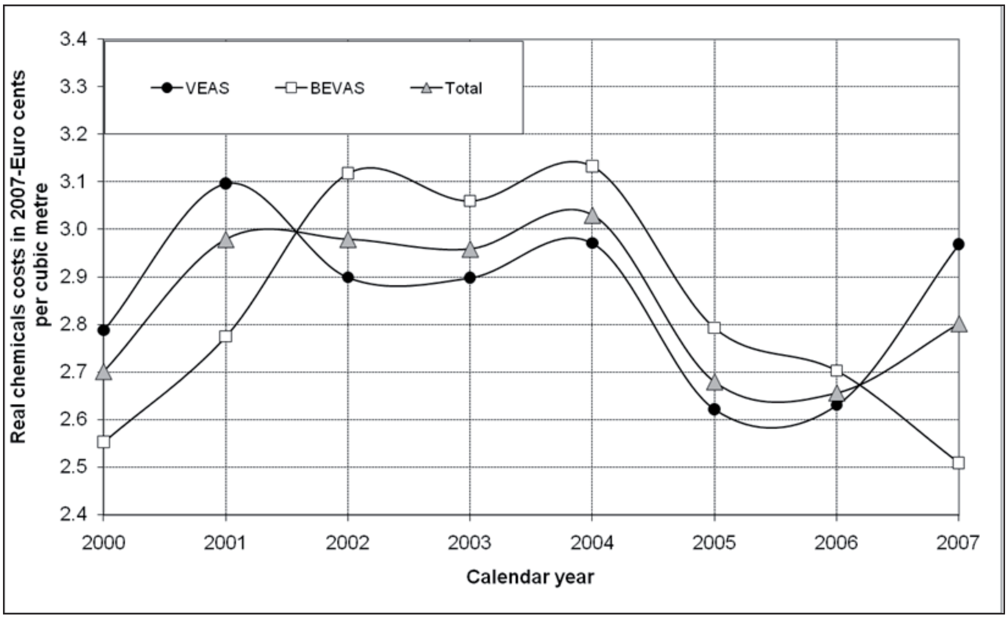

Figure 6: Specific real chemicals costs in Oslo's WWTPs

Comparative discussion

The WWTPs expended more on energy and chemicals per cubic metre of wastewater treated, as compared to what the WTPs spent on the same factors of treatment per cubic metre of water supplied. However, it should be borne in mind that the volume of the water treated and supplied is less than the volume of wastewater collected and treated (as the latter includes stormwater as well). If expressed on a per-capita-serviced-basis (common denominator), the WWTPs, as seen in *Figure 7*, spent over 8 Euros per year on chemicals and energy, with a maximum of 10 Euros in 2005. The WTPs on the other hand, spent less than 3 Euros per capita per year from 2000-2004, before the rise from 4 Euros to 8 Euros began in year-2007. For WTPs and WWTPs taken together, the rise was from 10 Euros in year-2000 to a little over 12 Euros in year-2004. It will be a safe conjecture if one can state that this would have increased to over 16 Euros in year-2009.

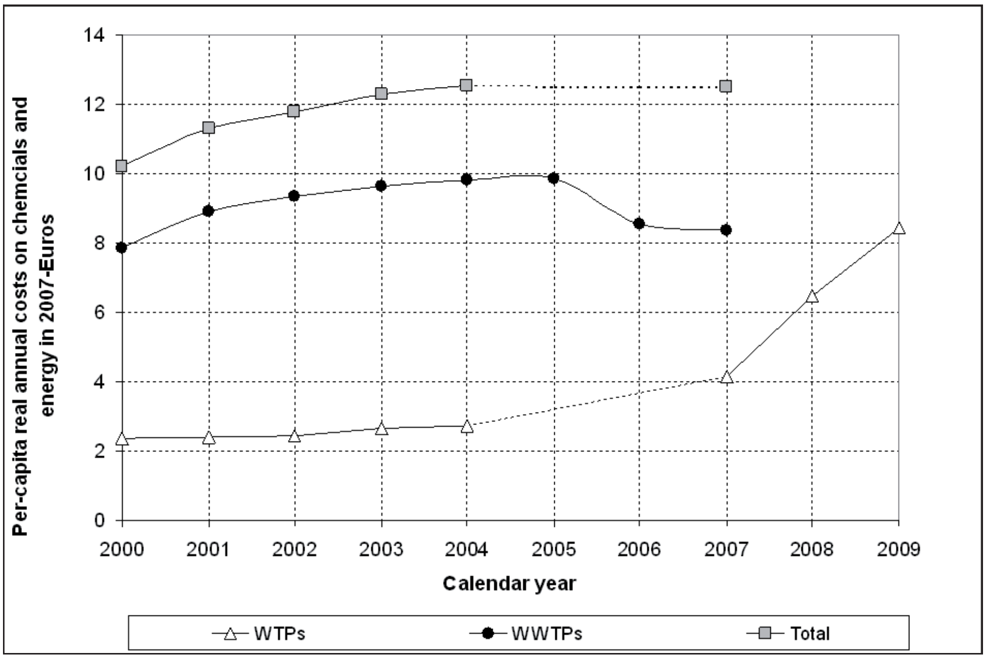

Figure 7: Per-capita real annual costs for chemicals and energy in Oslo's WTPs and WWTPs

Chemicals, it is seen, have cost the WWTPs more than energy (almost twice in year-2000 to nearly on par in 2005); while in the case of the WTPs, it has been the other way round. The upgrading in 2008-2009 has taken the ratio up to 0.5 in year-2009.

Measuring benefits of treatment

Provision of clean potable water and the treatment of wastewater prior to disposal may be social (and environmental) responsibilities of the operating agency – in this case, the Municipality of the city of Oslo. The direct benefits of providing clean water to the consumers and treating the wastewater discharged before it drains into the *fjord*, are difficult to express in monetary terms, though one can think about the impact the absence of such services would have on social welfare, human productivity, environmental upkeep and long-term detrimental consequences on the economy.

So the main benefit the consumers avail of, can be thought of in terms of the avoided expenditure on healthcare, for example. The economy of the city would benefit by a reduction in water-related-sickness-induced absenteeism. The benefits for the environment (the fjord in this case), for that matter, could be directly related to those for the people whose livelihoods depend on it – fishermen especially. Of course, one can distinguish between and include short-term, medium-term and long-term benefits. The complexity associated with such an exercise warrants a separate detailed investigation in another paper.

Here, as referred to earlier in the paper, shadow pricing in order to determine the benefits of wastewater treatment for wetlands restoration in the Mississippi river valley in the USA has been done (unpublished work referred to in Corcoran, E et al, 2010; referred *Appendix IV).* Efforts, on the part of the authors of this paper, to find out if similar studies have been carried out for a Scandinavian or West-European setting proved futile. The authors refrain from using the shadow pricing values referred to for the American setting for a cost-benefit analysis of wastewater treatment in Oslo.

CONCLUSIONS AND LIMITATIONS

In the preceding paragraphs, the cost aspect of energy and chemicals consumption in Oslo's WTPs and WWTPs was analysed. Specific real costs were calculated and compared over time to understand chemicals and energy consumption as cost drivers (cost elements in other words) in WTPs and WWTPs.

It was observed that chemicals, in general, cost more but caused less environmental impacts than energy in WWTPs, while for the WTPs, it was exactly the other way round. Real costs of energy and chemicals per cubic metre of fluid handled increased for WTPs during the 2000-2009 period – quite strikingly from 2007 to 2009. On the other hand, the corresponding specific expenses for WWTPs decreased after rising to a peak in year-2005. On a per-capita basis, the total specific energy and chemicals costs for WTPs and WWTPs taken together rose over time, gradually at first, till year-2007 and thereafter, one may surmise, the rise was rapid, thanks to the rapid increase in the specific costs at the WTPs.

Chemicals and energy accounted for between 16.9% and 19.7% of the total annual expenses at the WWTPs. As a percentage of the total O & M expenses alone, the share ranged between 46% and 53%. (*vis-à-vis* a maximum of 44% in Greece (Tsagarakis et al, 2003) and an average of

35% in the Scandinavian case study by Balmer (2000)). The differences can possibly be explained by the widely varying share of manpower costs in the O & M expenses. A point to note here is that while chemicals cost the Oslo WWTPs more than energy did, Balmer (2000) and Tsagakaris et al (2003) have reported the opposite in their respective case studies. The reason could be a higher degree of wastewater and sludge treatment in Oslo which necessitates a relatively greater consumption of chemicals, availability of relatively cheaper electricity for the Oslo WWTPs, and the avoidance of purchase of natural gas owing to use of internally-generated biogas for heating applications. Balmer (2000) has put the nominal average per-capita annual O & M expenses in Scandinavian WWTPs in 2000, at between 6-12 Euros. For the WWTPs in Oslo, this figure was 9.96 Euros – close to the midpoint of the range.

In the paper, some simplifications and assumptions were resorted to. The cost of combusting biogas and transporting the heat in BEVAS was neglected; and in VEAS, this was assumed to be part of the assumed cost of electricity generation from biogas. The assumption that the cost of generating biogas at VEAS is an average of the electricity tariffs for the period 2000-2007 is quite gross and needs to be corrected, as and when accurate data are available. As the focus was restricted to the actual direct expenses incurred by the plants, avoided expenses and revenue generated by the sale of by-products were not taken into consideration.

BEVAS did not (and does not at the time of writing) generate electricity from the biogas it produces. But this may change in the future, and there may even be revenue generation for the plant by the sale of the biogas to the transportation sector. If the Skullerud WTP can avail of a more reliable power supply in the future, the purchase of diesel may not be necessary. Further efficiency improvements and optimisation in the use of chemicals may bring down the per-unit-volume (specific in other words) consumption of energy and chemicals; and slightly offset the inevitable rise in prices.

If investments are directed at reducing the operational expenses, savings by way of optimised consumption of energy and chemicals at the treatment plants without impacting the service levels negatively, can be channelled into the former, in order to further upgrade performance. It should however be said that the scope for reducing costs is system-wide and is influenced by both inherent systemic lock-ins and external factors beyond the direct control of the utility.

Several scenarios or combinations of scenarios at the WTPs and WWTPs can be contemplated upon, and the effect of changes in the factors of water and wastewater treatment – both external and internal – on the costs (and specific costs) can be examined in further studies. Also of interest would be an attempt to develop a methodology to measure the benefits (monetary) of both water and wastewater treatment – by resorting to shadow pricing methods - and compare the same with the costs of the same.

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APPENDICES

Appendix I

Chemicals and energy consumed at Oslo's WTPs (Nominal price of chemicals in NOK per MT is given within parentheses below the mass)

	2000	2001	2002	2003	2004	2007	2008	2009
$Al_2(SO_4)_3$ (kg)	253300	358210	240000	205000	188680	205920	237500	152000
	(1403)	(1419)	(1419)	(1434)	(1485)	(1650)	(1651)	(1651)
$Ca(OH)_2$ (kg)	369100	436680	378000	358000	347860	335280	1655000	3056000
	(1089)	(1102)	(1102)	(1114)	(1153)	(1281)	(1281)	(1281)
CO_2 (kg)	350800	336238	320000	313000	325480	286000	1632000	3104000
	(731)	(740)	(740)	(748)	(774)	(860)	(860)	(860)
Cl_2 gas (kg)	48660	47514	53800	55341	56455	4000	0	0
	(10200)	(10320)	(10320)	(10440)	(10800)	(12000)		
Microsand (kg)	0	0	0	0	0	0	128000	281600
							(670)	(670)
PAX (kg)	0	0	0	0	0	0	1200000	2640000
							(1200)	(1200)
Polymer (kg)	1375	2125	1400	1850	1550	1430	21875	45200
(For Oset)							(26300)	(26300)
(For Skullerud)	(33405)	(33798)	(33798)	(34191)	(35370)	(39300)	(39300)	(40000)
NaOCl (kg)	0	0	0	0	0	342000	244500	268000
Electricity (GWh)	19.97	19.77	19.6	18.8	20.1	22.8	30.82	33.6
Diesel fuel (MT) (at Skullerud)	0	0	0	0	151.9	180.2	186.4	191.4

Masses of chemicals and energy consumed at Oslo's WWTPs

	2000	2001	2002	2003	2004	2005	2006	2007
Iron chloride (MT)	12648.0	4940.0	2939.0	3302.0	2810.0	2289.0	2464.0	2619.0
Iron sulphate(MT)	0.0	1324.0	2597.0	1760.0	1754.0	1816.0	1901.0	2519.0
PAX(MT)	4374.0	3164.0	2430.0	2070.0	2062.0	2018.0	2517.0	2552.0
Methanol(MT)	2083.9	2120.6	2179.1	2297.3	2048.2	2224.7	2304.5	2397.7
Ethanol(MT)	0.0	11.858	12.0	19.68	19.52	19.80	1.520	0.0
Nitric acid(MT)	1396.0	1464.0	1833.0	1833.0	1852.0	1729.0	1927.0	2040.0
Calcium hydroxide(MT)	2108.0	2502.0	2353.0	2240.0	2172.4	2344.0	2302.5	2349.0
Polymers(MT)	97.3	64.7	80.2	85.1	87.2	83.3	75.2	87.5
Electricity consumption(GWh)	30.97	34.25	37.49	37.22	37.82	38.70	38.99	39.17
Heating oil consumption (GWh)	1.70	1.43	1.16	0.89	1.39	0.99	1.21	1.11
Biogas-derived heat (GWh)	40.23	42.22	41.23	43.08	45.99	51.25	49.54	47.05

Appendix II

Consumer price indices and exchange rates

Calendar year	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009
CPI (1998=100)	105.5	108.7	110.1	112.8	113.3	115.1	117.7	118.6	123.1	127

(Courtesy Statistics Norway – www.ssb.no)

Year	EUR/NOK
2000	8.05
2001	8.00
2002	8.37
2003	8.00
2004	7.51
2005	8.04
2006	8.11
2007	8.66
2008	8.50
2009	8.39

(Courtesy Norges Bank, Norway – www.norges-bank.no)

Appendix III

Unit prices of energy elements (from Statistics Norway)

Calendar year	Heating oil (€-¢/kWh)	Diesel (€/litre)	Electricity (€-¢/kWh)	Natural gas (€-¢/kWh)
2000	8.20	1.21	4.34	2.24
2001	8.50	1.07	4.35	2.25
2002	7.17	0.96	4.40	2.03
2003	8.50	1.03	5.75	2.13
2004	9.59	1.12	5.57	2.40
2005	11.19	1.27	5.82	2.86
2006	10.48	1.32	4.73	3.21
2007	9.82	1.20	6.45	3.12
2008	10.24	1.13	7.60	3.88
2009	10.49	1.15	8.15	---

Appendix IV

Wastewater destination	Treated water (Euros per cubic metre)	Nitrogen (Euros per kg)	Phosphorus (Euros per kg)	Suspended solids (Euros per kg)	COD (Euros per kg)	BOD (Euros per kg)
Sea (Fjord)	0.1 Euros per cubic metre	4.61	7.53	0.001	0.01	0.005

Paper no. 8: ([130] in List of references)
R. Ugarelli, G. Venkatesh, H. Brattebø and S. Sægrov. Asset management for urban wastewater pipeline networks.
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Paper no. 9: ([131] in List of references)
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