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Stormwater quantity and quality in a multiple pond–wetland system: Flemingsbergsviken case study

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Abstract

This study aims at presenting a methodology for quantifying yearly and monthly material transport into and out of stormwater treatment facilities (STFs) and at presenting problems related to monitoring and estimation of reduction efficiency. The model developed uses a sub-watershed approach. It employs standard values of runoff coefficients and pollutant concentrations together with precipitation data and estimated areas of different land uses within sub-watersheds. Both standard concentrations and field sampling data show that the inflow water to the facilities at Flemingsbergsviken is nutrient rich. The metal concentrations of lead, copper and zinc are relatively low, but may nevertheless increase the risk for negative impacts on aquatic life. The monthly values of runoff coefficients, used to estimate the monthly runoff water flow, vary largely between years and months. This makes the estimation of flow uncertain. A comparison of standard concentrations with measurements shows good correlation for nitrogen. The standard concentrations of total phosphorus and metals were higher than most of the sampled concentrations. However, the sampled concentrations are uncertain since they have been sampled instantaneously. An improved strategy concerning monitoring is discussed. © 2000 Elsevier Science B.V. All rights reserved.

Keywords: Concentration; Material transport; Metals; Nutrients; Stormwater; Wet pond; Wetland

1. Introduction

Separate transport systems for stormwater (rain and melt water runoff) in the form of open ditches or stormwater sewers are used for conducting the water to open stormwater treatment facilities (STFs). STFs are 'ecotechnological' facilities that employ natural processes for the removal of pollutants and nutrients in stormwater. Examples of STFs are wet and dry ponds (retention and detention ponds), constructed wetlands, open ditches and green structures such as grassed swales, infiltration trenches and infiltration strips. The advantages of using STFs lies in their aesthetic values and in their pollutant treatment and flow compensation capabilities. The alternatives are to conduct the stormwater directly to, for example, lake recipients or to lead the water to wastewater treatment plants. These are not sustainable alternatives, the former creating negative impacts in the recipient and the latter increasing the metal content of the sludge (preferably to be

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reused in agriculture) and leading to intermittent loadings disturbing the reduction efficiency of the plant. In many cases, the alternatives also lead to increased need of investments in sewer systems.

In Sweden, particularly during the last couple of years, there is a rapidly increasing trend of using STFs. However, there are problems that need to be further investigated and solved. So far, we only have limited experience concerning the function and reduction efficiency of different STFs, single and in combinations. Furthermore, there is a need for more rigorous design criteria for Swedish conditions and tools for making proper choices of treatment methods. The location of the STFs should also take site specific conditions into consideration. At a recent conference (VAV-dagen, June 1997) two out of the three research topics which were set as priorities were related to stormwater management. These were the following-up of existing STFs and recommendations for design and maintenance of STFs (Malmqvist, 1997). These problems and proposed research areas were also stressed at the international conference Stormwater Management—Creating Sustainable Urban Water Resources for the 21st Century held in Malmö, 7–12 September, 1997.

The stormwater treatment constructions at Flemingsbergsviken were finished in May 1995, in an urbanized watershed to Lake Orlången near Stockholm, and compose the basis for our studies. The facility has been in operation since 1995 and it is now, some 3–4 years after the construction, that these types of facilities usually prove to be effective. Negative nutrient reduction efficiencies of −33% (total phosphorus, tot-P) and −9% were, for example, estimated for the combined wet ponds and constructed wetland facility in Toftanäs, near Malmö in Sweden, during the first year of operation. These values increased to 31% and 35% respectively after 3 years (Stahre and Larsson, 1993).

The monitoring programme provides a possibility of studying separate STFs and different combinations regarding seasonal variations of water quality and quantity. There are several treatment steps used in the large facility which makes the facility unique: a pond with oil separation, wet ponds, constructed wetlands and open ditches (Figs. 2 and 3). The purpose of the STFs is to improve the condition of Lake Orlången. The lake is eutrophic and it consists of sediments that are contaminated with metals.

A number of models are available for quantifying urban runoff, including water quality. Most of these models express the material loading, i.e. the pollutant load rate, in terms of, for example, kilograms per year (Nix, 1994). The models differ in complexity and need different types and amounts of input data. Typically, steady-state models are usually simpler than dynamic or transient models. The simpler models require little input data and little or no help from a computer. Their main purpose is to calculate long-term averages of, for example, annual runoff volumes and pollutant loads (Nix, 1994) or to set up simple water and mass balances. They are useful for locating problem areas, i.e. they are used as a basis for deciding where to implement different types of STFs. Examples include SWMM Level I (Nix, 1994) and the Simple Method (Schueler, 1987). These models quantify pollutant loads from runoff coefficients, concentration data, site area and rainfall depth. The Simple Method extends to use a factor correcting the rainfall data for storms producing no runoff. However, the Simple Method does not consider base flow runoff and associated pollutant loads, and is better used at small development sites (Schueler, 1987; Andrews, 1992).

The more detailed models, such as STORM, SWMM, Mouse, XP-AQUALM and Walker's model (P8), are on the other hand mainly computer executed and generate hydrographs (flow versus time) and pollutographs (concentration versus time) at one or a few points in a watershed (STORM) or at various locations (SWMM). These are capable of simulating watershed behaviour over long periods by continuous simulation. For example, the effects of several consecutive rain events may be simulated. STORM also accounts for precipitation held in surface depressions, i.e. not appearing as runoff (Nix, 1994). These provide more detailed descriptions of the watershed, but are relatively complex if considering the often limited amount of data available for calibration in watersheds and sites of STFs, especially regarding yearly periods. For example, P8 requires data on soil types and continuous hourly precipitation records. According to Wanielista and Yousef (1993) there prevails large variability of nutrient and metal loadings from one location to another. Therefore, it is important to carry out site specific studies to estimate pollutant loadings.

In this study we summarize and interpret data from a particular stormwater treatment facility in Sweden and discuss issues regarding the functionality and design. The presented data include unique long-term and winter-time data. Specific objectives of this paper are (1) to present a methodology for material transport calculation to stormwater facilities, (2) to discuss limitations and problems regarding monitoring and function of the facilities and (3) to present the stormwater compounds from different land uses. This study starts with presenting the different treatment steps of the facility and the monitoring programme. Then it describes and quantifies hydrological characteristics regarding precipitation, runoff coefficients and flow on a yearly and monthly basis (i.e. water quantity). Water quality is presented as sampled and standard concentrations. These concentrations are also compared with stormwater criteria and used for quantifying material transport of nutrients and metals. Furthermore, factors that are related to the estimation of reduction efficiency are discussed. A model (referred to as Stormtac) has been developed and used for the calculations in this study.

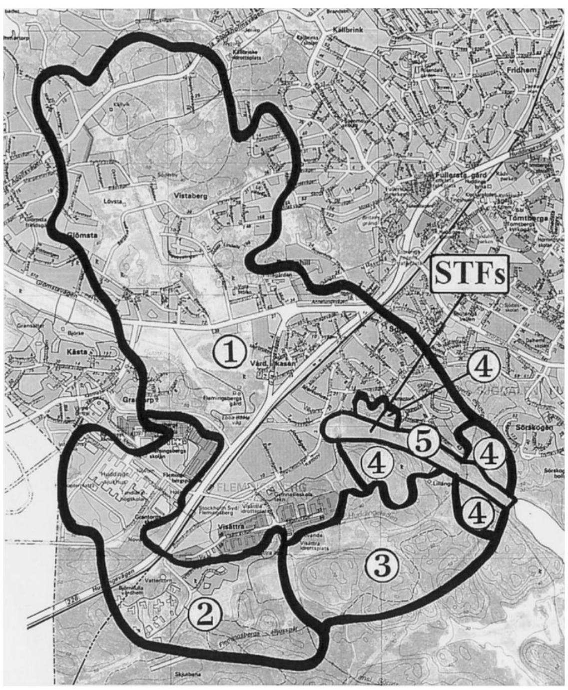


Fig. 1. Watershed area divided into sub-watershed Areas 1–5.

2. General description of the facility

Lake Orlången is a eutrophic lake and recipient for the stormwater from the studied Stormwater Treatment Facilities (STFs). Phosphorus is generally a limiting factor for plants in fresh water lakes. It is therefore especially important to reduce phosphorus loadings to the lake. However, in eutrophic lakes nitrogen (as in the seas) can be limiting to growth during some periods. Therefore, it is also important to reduce nitrogen loadings. Furthermore, the sediment content of metals is also rather high in Flemingsbergsviken, which is the bay recipient of the water from the facilities. The watershed connected to the STFs is furthermore the most urbanized sub-watershed in the lake watershed. The lake is a source lake and therefore any improvement of the lake water quality also will lead to improvements in the other lakes further down in the same lake system.

The STFs are situated 10 km south of Stockholm and take up an area of 18 ha (length 1200 m, width 150 m), i.e. around 0.2 km^2. This land area was before the construction an open field with ditches transporting the polluted stormwater relatively fast into the lake recipient. The watershed area (Fig. 1) that is connected to the facilities is estimated to 9.6 km^2 to be compared with the watershed of the whole lake Orlången of 40 km^2 and the lake area 2.6 km^2. The studied watershed area has the following land use distribution: 52% forest, 25% houses, 9% parks, 5% apartments, 3% industries, 3% other urban areas (such as commercial buildings and hospitals) and 1% larger roads, i.e. around 50% urban and 50% rural ar-

Fig. 2. Aerial photo of the stormwater treatment facilities at Flemingsbersviken.

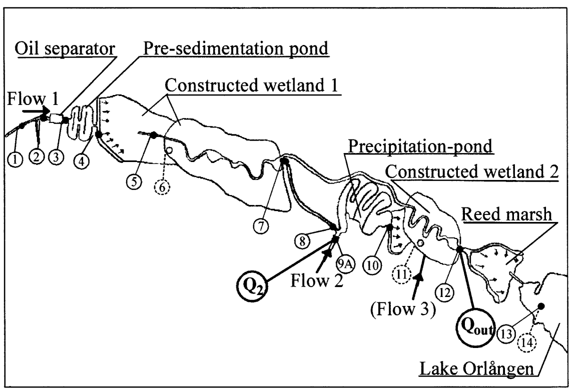


Fig. 3. Stormwater treatment facilities and points of flow measurements (Q_2 and Q_{out}), sediment sampling points (6, 11 and 14) and points of water samples (the other numbered points). Pollutant concentrations in the inflow and the outflow of each treatment step are analysed from the water samples. The three largest point inflows to the facility (Flows 1–3) are also shown.

eas. The STFs take up 2% of the watershed (reckoned from the outlet of the facilities). The polluted stormwater is cleaned and flows retarded towards Lake Orlången through natural ('eco-technological') facilities such as wet ponds, constructed wetlands and open ditches. These are not lined facilities, and submerged growth is involved.

The STFs have three point inflows (Fig. 3). The first inflow (Flow 1) receives stormwater from the northern part of the watershed (Area 1, Fig. 1) by an open ditch. The water first passes a coarse grid and an oil separation pond using a floating flange (Figs. 2 and 3). It has an area of 630 m^2 and a volume of 660 m^3. The surface loading on this facility is 0.4 m/h at the dimensioning flow (Q_{dim}) (see Section 2.1). Then the water continues to a meandering pre-sedimentation wet pond: a pond with permanent water level (2900 m^2, 3800 m^3, surface loading 0.09 m/h at Q_{dim}, sedimentation time 12 h and detention time 15 h at Q_{dim} (Signeul, 1996). The meandering form created by using baffles or islands within the pool can reduce short circuiting, increase the flow path length (Water Environment Federation and American Society of Civil Engineers, 1998) and thereby increase the vegetation contact area and improve the efficiency of reduction processes. In these steps the water is above all cleaned from oil, grease, coarse material/sediments and particle bound pollutants and nutrients. The water is spread to the following constructed wetland by a distribution ditch in order to avoid the creation of a straight channel flow across the wetland.

The first step after the distribution ditch consists of a filter strip (the first part of Constructed Wetland 1; see Fig. 3) of around 7500 m^2. The constructed wetland part (2 ha, around 14 000 m^3 and detention time 2.3 days at Q_{dim}) has a water depth varying between 0 and 1.9 m. It treats the water by processes such as plant uptake (biosorption) of soluble pollutants, sedimentation, adsorption and denitrification. An outlet wet pond is situated at the end of the wetland, after which the water is conducted in an open ditch to be mixed with the other inflow (Flow 2) from Area 2. There is also an overflow after the outlet pond which is in operation when the flow exceeds 60 l/s. This overflow bypasses the precipitation pond for Flow 2 in an open ditch but is further treated in the following constructed wetland.

Flow 2 is conducted to the facility by a stormwater tunnel and a flow regulator. The flow is limited to maximum 60 l/s by the regulator. At larger flows, the water is stored in a basin at the tunnel (Lännergren, 1998). At the outlet of this tunnel the precipitation chemical PAX-XL60 (poly aluminium chloride) is added in the amount of 20 ml/m^3, of a 7% water solution, to a mixing basin for Flow 1 and 2. The time for this addition is late autumn and early spring. The purpose with this is to secure a more effective phosphorus reduction during this period of time. The phosphorus is in this way precipitated to the sediments in the following precipitation pond (10 000 m^2, 15 000 m^3, depth 1.5–2.0 m, surface loading 0.04 m/h at Q_{dim}, sedimentation time around 25 h and detention time 1.5 days). However, no consistent positive effect of the precipitation has been observed (Lännergren, 1998). In this meandering wet pond, sedimentation also occurs and the sediments are to be taken away by suction at a few years interval (which also will be done in the pre-sedimentation pond of Flow 1).

After the pond, the water is distributed to another constructed wetland part (1 ha, 7500 m^3 and detention time around 1 day at Q_{dim}) with the water depth 0–1.5 m and with an outlet pond, i.e. similar steps as for Flow 1. Finally, the flow is led to Lake Orlången through a common reed marsh. Flow 3 from Area 3 is a smaller point flow from, for example, Visättra sports ground, led to the constructed wetland (located after the precipitation pond) by an open ditch. There are also diffuse surface water flows (Flow 4) from the surrounding grounds (Area 4) of the facility; this flow is relatively small (Table 3). The net atmospheric fallout (precipitation–evapotranspiration) on the facility area (Area 5) is accounted for by Flow 5, also small.

The estimated average flow is 1.2 Mm^3/year, fluctuating from 0.9 to 1.6 Mm^3/year for the reference period of 1961–1990. The average outflow from the facility has accordingly been estimated to approximately 3400 m^3/day. Earlier estimations were approximately 1.8 Mm^3/year. The difference can be explained by decreased runoff coefficients used in the model, especially for forests (from the value 0.3 to 0.1).

2.1. Design criteria

The dimensioning stormwater fluxes from the different sub-watersheds of Flows 1–3 have been estimated using the design 2 year rain (rain duration and intensity statistics for Stockholm), water velocity, runoff coefficients and land use areas. The dimensioning fluxes, i.e. the hydraulic loadings, were calculated at 2300 l/s (Flow 1), 1500 l/s (Flow 2) and 50 l/s (Flow 3). These values have been used for the design of the required flow compensation volumes, e.g. in the tunnel basin before the precipitation pond.

The different treatment units have been designed using a surface loading criteria with in data consisting of chosen sink velocities of particles together with a dimensioning flow (Q_{dim}) of two times the yearly average flow (in data: yearly precipitation, runoff coefficients and land use area). The yearly average flow was estimated at 35 l/s for Flow 1, 24 l/s for Flow 2, 6 l/s for Flow 3 and accordingly 65 l/s for the outflow. The required area of the ponds (A_{pond}) can be calculated from Eq. (1):

$$A_{\mathrm{pond}} = \frac{Q_{\mathrm{dim}} \times 0.001 \times 3600}{v} \tag{1}$$

where A_{pond} is required pond area (m^2), Q_{dim} is dimensioning flow (l/s) and v is particle sink velocity (m/h).

The precipitation pond is for example designed for the flow 118 l/s and the particle sink velocity 0.04 m/h ($A_{\mathrm{pond}} = 118 \times 0.001 \times 3600/0.04 = 10\,600$ m^2). The volume is estimated by choosing a suitable pond water depth (1–2 m). The water residence time of Flow 1 through the facility is estimated to >5.5 days and >2.5 days for Flow 2 (Signeul, 1996). The two ponds at Flemingsbergsviken were designed for a sink velocity of 0.04–0.09 m/h. A sink velocity of 0.07 m/h corresponds to a particle size of around 5 μm. A study has shown that around 90% of the particles in stormwater are within the diameters 10–35 μm and that almost all are within 5–45 μm (Urbonas and Stahre, 1993).

3. Monitoring programme

Monitoring data (sampling concentrations and measuring flow) exist since October 1995. There are points of sampling before and after each specific treatment step (Fig. 3). It is difficult and very expensive to accurately monitor stormwater quality and quantity at all these points. Sampling of concentrations without flow measurements are for example not recommended for reliable transport estimations. The chosen strategy for these facilities was to continuously register flow at one inflow and the outflow, and to take instantaneous spot samples during a time period exceeding three years to make a study of seasonal variations and of yearly reduction efficiency possible. Complementary and more intense sampling have also been carried out during some time periods. Experiences during the first years will lead to revisions of the monitoring programme.

The programme (Gunsell, 1995) includes the following data: flow, precipitation, water and air temperature, pH, conductivity, bacteria and concentrations of phosphorus (tot-P, PO_4-P), nitrogen (tot-N, NO_2-N, NO_3-N, NH_4-N), SS, DS (dry substance), Cl, BOD_7, COD, TOC and metals (Hg, Cd, Pb, Cu, Zn, Cr, Ni, Fe, Mn and Al). The metal analyses have been performed on unfiltered samples providing both soluble and particle bound fractions (Lännergren, 1998), i.e. total metals. Precipitation is continuously recorded at the tunnel (point 9A, Flow 2). Flow is measured by continuously recording indicators at the tunnel (point 9A, Flow 2) and at the outlet (point 12) (Fig. 3). The indicators, type Cerlic DLF-μP, are placed in pipes upstream of straight weirs and register water level and velocity. The flow is automatically calculated from this data. However, complementary flow measurements are desired for a more accurate quantification of the different flow paths. Unfortunately, continuous flow measurements are expensive. Flow 1 is not measured due to economical reasons and problems of finding a location for representable flow measurements (there are, for example, very low altitude differences from one end to the other of the facility; only 60 cm). Flow 3 is not measured due to economical reasons and since the flow is relatively small.

The following parameters are sampled each month (one measurement per month): tot-P, tot-N, water and air temperature, pH and conductivity. The water sampling frequency of all water related parameters mentioned is otherwise four times per year. Complementary more intense sampling has been carried out during May 1997 (Stark and Witte, 1997). Sediment samples (tot-P, PO_4-P, tot-N, DS and the metals mentioned above except Al) are also taken from within the facilities (one sample per year at the depth 0–1 cm; Lännergren, 1998). The accumulated depth of the sediments is measured approximately once per year. The sampling and the analyses of temperature, pH and conductivity were performed by Yoldia Naturundersökningar (Huononen, 1997) whereas the nutrient and metal analyses were performed by Scandiaconsult Miljöteknik in Malmö, Sweden. A spectrometer was used for the analyses of tot-P (SS 028127-2) and tot-N (SS 028131-1). Flame atomic absorption spectrometry (SS 028152-2, SS 028150-2) was used for the metals.

Recipient data (sampling in the bay recipient near the outlet of the facility) are measured twice per year. The following recipient data are included in the monitoring programme: temperature, pH, conductivity, Cl, TOC, tot-P, PO_4-P, tot-N, *Escherichia coli*, transparency, O_2 and chlorophyll *a*. A maintenance programme is also in operation (harvesting of vegetation each autumn and planned sediment removal). The purpose is to prevent the nutrients and metals, which have been taken up by the plants, from being released and transported out from the facility when the plant material has been mineralized. According to the monitoring programme ground water quality is to be analysed in existing pipes. However, such analyses have not been made because the existing pipes are made of metal that may contaminate the samples. In addition, the water level in the pipes is so high that analyses were not considered to add any information about the groundwater in the area (Lännergren, 1998). These pipes should be fixed so that quantitative and qualitative ground water measurements may be studied. The ground water is assumed not to be affected considerably as the ground is of dense clay.

In this paper the following parameters will be focused upon: total phosphorus (tot-P), total nitrogen (tot-N), total zinc (Zn), lead (Pb) and copper (Cu). These nutrients and metals have been chosen since they may have large effect on the lake recipient and since they are among the parameters that we have most information on (the estimated standard concentrations are consequently more reliable).

4. Hydrological characteristics

4.1. Precipitation

Both a local station and the Swedish Meteorological and Hydrological Institute precipitation gauge in Stockholm have been used in order to estimate the yearly precipitation (p), i.e. both rainfall and snowfall. The station 'Stockholm, Observatorielunden' is situated about 15 km north of Orlången and has been chosen as the reference station for the watershed studied, to be used when local data are missing and for comparisons (Fig. 4).

The precipitation values from SMHI from 1995 to 1997 have been used in the flow calculations since these values exist for the whole study period and since they compare well with the values from the local station (Fig. 4).

The precipitation values in Table 1 are taken from the SMHI station Stockholm (Swedish Meteorological and Hydrological Institute, 1996) and are adjusted for sampling errors (wind influence, evaporation and adhesion) by using the correction factor 1.15 for the transformation of sampled precipitation data to real (true) precipitation (Larm, 1997). In addition to uncertainties due to this adjustment there are also errors depending on the localization and number of rain gauges. The studied year of 1997 had approximately the same precipitation intensity (0.61 m/year) as during the average year in the reference period 1961–1990, a period with an average precipitation of 0.62 m/year. The year 1996 was extremely dry (0.43 m/year); similar to the precipitation intensity during the minimum year in the reference period.

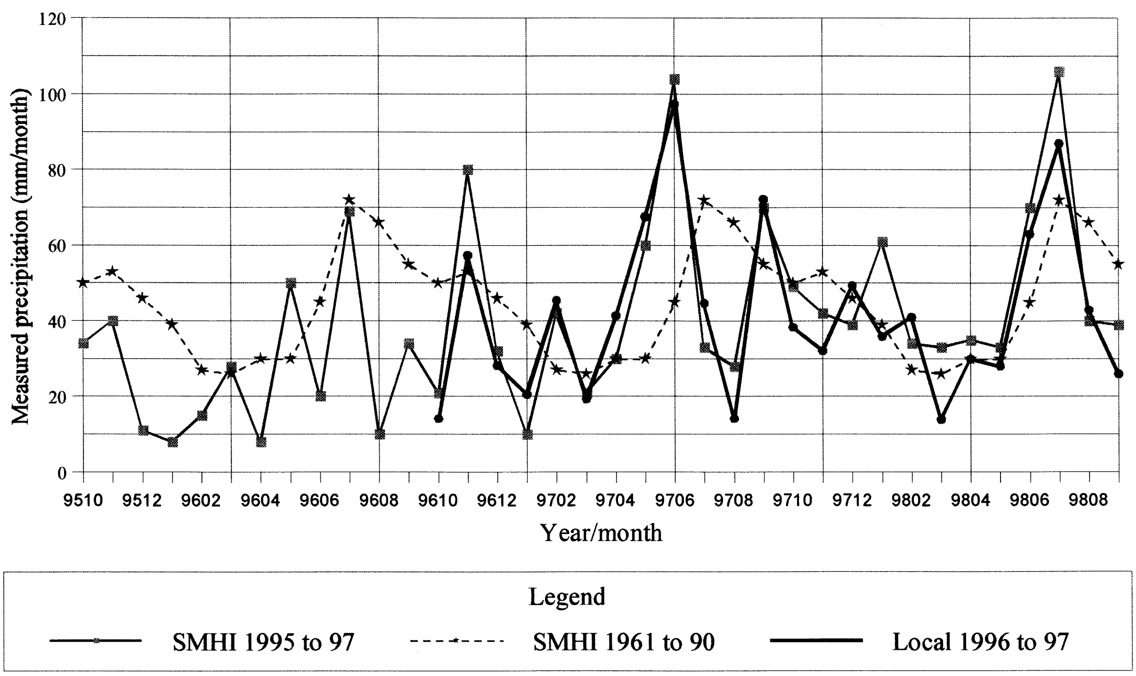


Fig. 4. Measured precipitation data (mm/month) from the SMHI-station Stockholm and from the local station within the facility area. The trace labeled SMHI 1961–90 represents the monthly averages for the reference period 1961–1990.

Table 1
Total land use distribution (km^2) in sub-areas A_j (where $j = 1,2,\ldots 5$ is the number of subareas), estimated yearly averaged runoff coefficients (φ) and calculated flow (Q)

Land use	A1	A2	A3	A4	A5	A	φ	Q ($\times 1000$ m^3/year)
Roads	0.088	0.018	0	0	0	0.106	0.85	56
Commercial areas	0	0.24	0	0	0	0.24	0.70	104
Industries	0.3	0	0	0	0	0.3	0.60	112
Apartment houses	0.39	0.12	0	0	0	0.51	0.48	152
Houses	2.2	0.22	0	0	0	2.42	0.25	375
Parks	0.44	0.28	0	0.1	0	0.82	0.18	92
Forests	2.54	0.91	1.2	0.33	0	4.98	0.10	309
STF land	0	0	0	0	0.147	0.147	0.20	18
STF water	0	0	0	0	0.033	0.033	1.0	0.3
Total	6.0	1.8	1.2	0.43	0.18	9.6	0.21	1217

4.2. Runoff coefficients

The runoff coefficient (φ) can be defined as the ratio between the runoff depth and the rain depth. The runoff coefficient expresses how much precipitation will become runoff after losses such as evapotranspiration (evaporation + transpiration), surface storage, infiltration and interception (precipitation adsorbed to the vegetation). First, runoff coefficients have been estimated from literature studies for the identified land uses within the watershed (Larm, 1996). These are compiled in Table 1 as yearly average values. However, the coefficients also vary in time; a higher coefficient could for instance be expected during winter than during summer since the infiltration capacity is lower during winter. However, during wintertime snow periods a storage occurs which can be accounted for by decreasing the runoff coefficient, whereas during snow melt the coefficient should increase. Therefore, it is difficult to estimate the wintertime runoff coefficient without flow data.

Monthly values of runoff coefficients have been estimated from existing flow data, precipitation data and estimated watershed area. Data from two case studies have been used in the model; the watershed MA18 to Lake Magelungen, further down in the same lake system and a sub watershed to Lake Trekanten, around 10 km from Flemingsbergsviken. Fig. 5 presents these values (median, mean and S.D.). Each monthly model value for Flemingsbergsviken (φ_{mF}) has been estimated by multiplying the monthly model value from references (φ_{mR}) with the fraction of yearly estimated runoff coefficient from Flemingsbergsviken (φ_{yF}) and ditto from the model value (φ_{yR}); see Eq. (Eq. (2):

$$\varphi_{mF} = \frac{\varphi_{mR}\varphi_{yF}}{\varphi_{yR}} \tag{2}$$

The median values (represented by the line in Fig. 5) show relatively small deviation between different months. However, the mean runoff coefficients are largest during the snow melt period around Mars and April.

The model values are compared to runoff coefficients estimated from outflow data at Flemingsbergsviken, using local precipitation data when available, or otherwise SMHI precipitation data. The four values with the largest deviation from model data were those based on SMHI data, i.e. the best comparison occurs for local precipitation data, see Fig. 5. The runoff coefficient can in this case

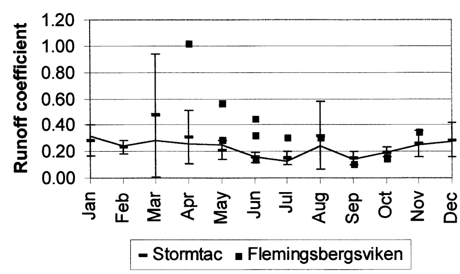


Fig. 5. Estimated monthly runoff coefficients in the model Stormtac (median = line, mean = — and S.D.) and ditto from measured data of Flemingsbergsviken (black squares).

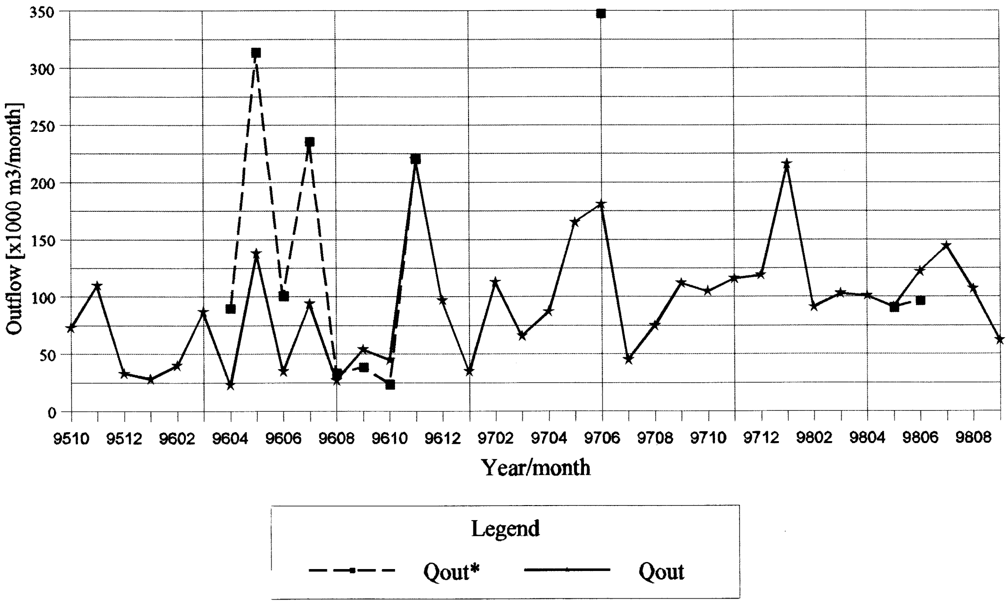


Fig. 6. Outflow from the facility from measurements (Q^*_{out}) and calculations (Q_{out}).

exceed the value 1 since it includes a storage factor of, for example, accumulated snow to be melted and to contribute to a large runoff during snowmelt in spring. The snow melts during different months different years. In this simplified calculation procedure temperature changes between years are not considered.

4.3. Flow

There are no flow measurements during the winter. Furthermore, existing flow measurements have relatively large uncertainties. Leakage of water passed the outflow indicator has occasionally been noticed, i.e. not all water has been passing through the point. This problem has been taken care off, at least partly. This indicates that the 'real flow' is higher than the measured flow (Q^*). Another uncertainty factor is the low difference in water levels. The straight weirs with large widths and shallow depths are also associated with large uncertainties. Furthermore, during one or a few occasions, a backflow has been reported from the lake to the facility which probably has been registered as an ordinary outflow. This problem has been solved by adjusting water levels in this part of the facility. Another problem observed is that thunder has lead to several periods without any flow data being recorded, e.g. during the period from July to September 1998. Flow calculations are necessary due to these problems and uncertainties.

Monthly flow values are needed for comparisons to the measured values, for estimations of flow-proportional concentrations and monthly loads. The calculated flow (Q), presented in Fig. 6, has been estimated from monthly specific runoff coefficients (median values in Fig. 5, calculated from Eq. (2)) and monthly precipitation data from SMHI. The following formula has been used for quantification of runoff water flow:

$$Q = p \sum \varphi_i A_i \tag{3}$$

where Q is runoff water flow (m^3/year or m^3/month), φ_i is yearly or monthly runoff coefficient for land use ($i = 1,2,\ldots N$), p is precipitation intensity (m/year or m/month) A_i is size (m^2) of land use ($i = 1,2,\ldots N$).

Q is used in the calculations of material transport (see Eq. (5)). The calculated annual flow corresponds to 0.13×10^6 m^3/km^2/year and has been verified to normal yearly values for this part of Sweden, estimated at 0.23×10^6 $m^3/km^2/year$ (Lännergren, 1993). The latter corresponds to a runoff coefficient of 0.37 which is higher than the runoff coefficient 0.21 of the studied watershed.

Estimations of sub-watershed areas, specific for each point of measurement and separated into different land uses, have been carried out by studying topographic maps and the technical storm water sewer systems within the areas. The flow values in Tables 1 and 2 have been derived using Eq. (3). For example, the flow from roads in Table 1 has been calculated as follows: $Q_{roads} = p \times \varphi \times A = 0.62 \times 0.85 \times 0.106 \times 10^6 = 55\,862$ $m^3/year$. The calculated flow from the water area of the STF (i.e. the net precipitation/evaporation) in Table 1 is estimated at only 330 $m^3/year$ (20 460 $m^3/year$ in as precipitation and 20 130 $m^3/year$ out as evaporation). The potential evaporation $E = 0.61$ m used was estimated from Penmans equation for the rain station Bromma/Stockholm during 1961–1978 (Bergström, 1993). This equation is based on energy balance with air temperature, solar radiation, air humidity and wind velocity as input data. This implies that the flow contribution from the water areas within the pond can be neglected on a yearly basis. The net precipitation/evapotranspiration within the STF (Area 5) is estimated at 18 000 $m^3/year$, which is smaller than the estimated diffuse runoff 32 000 $m^3/year$ from the surrounding grounds of the facility (Area 4) (see Table 2). Generally, groundwater inflow and outflow within the STFs may affect the outflows, these variables vary largely depending on site specific (e.g. hydrological) conditions. Here, the net contribution to/from the groundwater within the STF area is assumed to equal zero. This assumption was also made by, for instance, Dellien and Wedding (1997) after simultaneous measurements of inflow and outflow of a pond with clay bottom. The diffuse flows from net precipitation–evapotranspiration and from the surrounding grounds to the facility was estimated to only 4% of the total water inflow.

Table 2
Area (A), runoff coefficients (φ) and calculated flow (Q) from different sub-areas

Area	A (km^2)	φ	Q ($\times 1000$ $m^3/year$)
Area 1	6.0	0.22	822
Area 2	1.8	0.24	271
Area 3	1.2	0.10	74
Area 4	0.43	0.12	32
Area 5	0.18	0.16	18
Total	9.6	0.21	1217

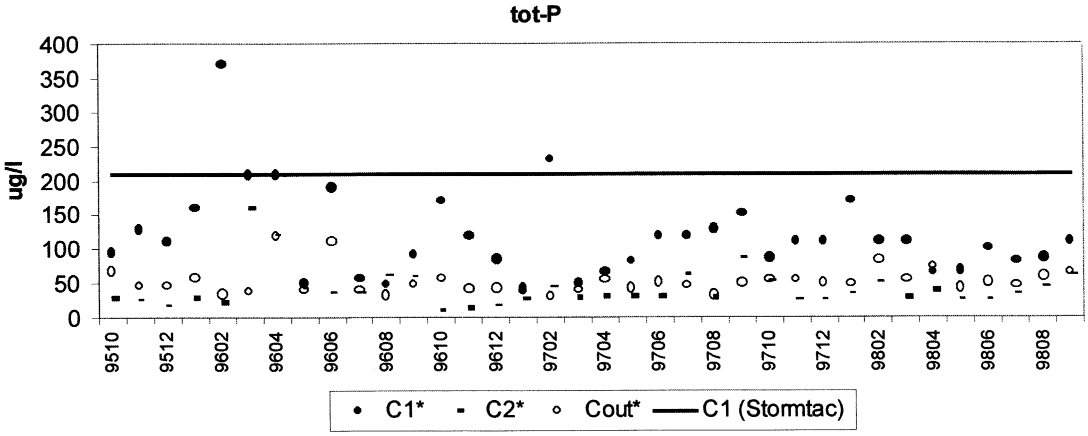
The water flows to the studied facilities have only been measured at point 9A (Flow 2, Q_2) and at the outflow point (Q_{out}). However, Flow 2 includes part of Flow 1 (the flow not exceeding 60 l/s; the rest is overflow), coming in by the open ditch (see Fig. 3). The proportion between Flow 1 and Flow 2 in the mixing storage basin at the tunnel is therefore unknown. The fact that these two latter fluxes are not separately measured has created a problem. The use of flow data from point 9A is therefore limited and the different inflows to the facility can better be estimated in relation to the outflow (Q_{out}). The outflow (see Fig. 6) is measured to be able to estimate the non-point discharges (Flows 4 and 5), but also for estimating Flow 1. The values from Table 2 give the flow from Area 1 as $Q_1 = Q_{out} \times 822 \times 1217 = 0.675 Q_{out}$. In the same manner $Q_2 = 0.223 Q_{out}$, $Q_3 = 0.061 Q_{out}$, $Q_4 = 0.026 Q_{out}$ and $Q_5 = 0.015 Q_{out}$.

A comparison of model calculated outflow versus measured outflow (Fig. 6) shows good correlation for some months and worse for others. However, no evaluation of the flow sub-model can be made since the measured values are very uncertain for this case study. When comparing flow data and precipitation (Fig. 6 versus Fig. 4) high flow is correlated to high precipitation and vice versa during each of the studied months.

5. Water quality

5.1. Measurements

Some of the measured nutrient (tot-P and tot-N) and metal (Pb, Cu and Zn) inflow concentrations (C^*) are shown in Figs. 7 and 8, representing a time period of over 3 years and points of measurements of Flow 1 (point 2) and Flow 2 (point 9A). Sampling in the inflow water exist only for Flow 1 (C_1^*) and Flow 2 (C_2^*). The outflow (C_{out}^*) has also been sampled. In average, the concentrations of tot-P and tot-N (Fig. 7) have been highest for Flow



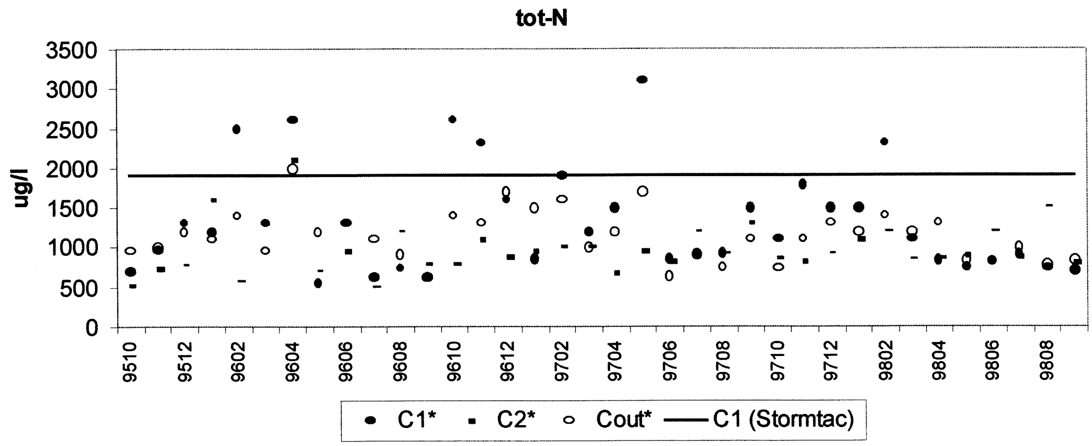


Fig. 7. Sampled concentrations in μg/l for Flow 1 (C_1^*), Flow 2 (C_2^*) and the outflow (C_{out}^*) of tot-P and tot-N for the stormwater treatment facilities of Flemingsbergsviken for the years 1995–1998.

1, lower in the outflow and lowest for Flow 2 (Lännergren, 1998). The analysed total nitrogen concentration was on average 15 (Flow 1) and 23 (Flow 2) times higher than the total phosphorus concentration during 1997. The median concentration value of tot-P is 110 μg/l for Flow 1 (the largest inflow). The corresponding value for phosphate phosphorus (PO_4-P) is 47 μg/l, i.e. the share of soluble phosphorus is around 43%. It is the phosphate phosphorus that has the largest impact in lake eutrophication processes. The median value of tot-N is 1150 μg/l. The corresponding value for the sum of nitrate and nitrite nitrogen is 370 μg/l (32% of tot-N) and for ammonia nitrogen only 25 μg/l (2% of tot-N). The soluble nitrate nitrogen, available for plant uptake, corresponds to one-third of tot-N. The sampled metal concentrations (Fig. 8) are briefly discussed in Section 5.3. Since a complete set of yearly flow values are missing and there are only instantaneous concentration values for this case study, flow-weighted concentrations have not been estimated from measurements and rely preferably on standard values.

5.2. Standard concentrations

The standard concentrations (C) are based on a rather comprehensive literature study (Larm, 1997). The standard concentration values for different substances and different land uses (from

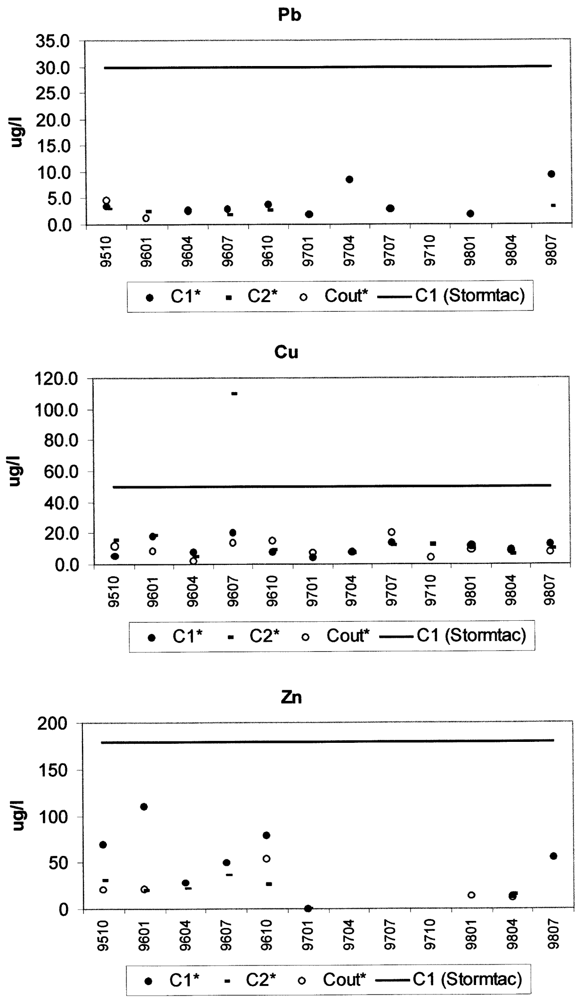


Fig. 8. Sampled concentrations in μg/l for Flow 1 (C_1^*), Flow 2 (C_2^*) and the outflow (C_{out}^*) of Pb, Cu and Zn for the stormwater treatment facilities of Flemingsbergsviken for the years 1995–1998.

literature) are continuously being revised and updated to include complementary land uses. The choice of standard values is correlated to how dense or spread out the specific land use area is in relation to the ‘average’ land use area density. Table 3 presents the up-to-date chosen standard concentrations for the land uses prevailing in the case study. In this case the values in the row of STFs refer to pollutant concentrations in atmospheric fallout (directly on the facilities). The ‘flow-proportional’ inflow concentrations from each sub-watershed area (Table 3, right part) have been estimated from calculated flow values (Q_i) and standard concentrations (C_i):

$$C = \frac{\sum Q_i C_i}{\sum Q_i} \tag{4}$$

where i is land use ($i = 1,2,\ldots N$). Instantaneous concentrations of pollutants in stormwater during a runoff event are highly variable and dependent on rainfall intensity, duration and the pollutant amount accumulated on the runoff surfaces. However, it may be preferable to use standard values instead of values based on instantaneous point measurements since the latter are not representative for larger areas and for longer time periods. It is also very expensive and time consuming to carry out continuous measurements at all representative points.

In the US Environmental Protection Agency’s Nationwide Urban Runoff Survey (NURP), significant differences in event mean concentrations were not detected among the urban categories

Table 3
Preliminary standard nutrient and metal concentrations (mg/l) for land uses in the case study (left) and estimated inflow concentrations from the different inflow areas (right)

Land use	C(P)	C(N)	C(Pb)	C(Cu)	C(Zn)	Area	C(P)	C(N)	C(Pb)	C(Cu)	C(Zn)
Forests	0.03	1.0	0.01	0.01	0.02	Area 1	0.21	1.9	0.03	0.05	0.18
Parks	0.08	7.0	0.02	0.02	0.02	Area 2	0.25	2.4	0.04	0.04	0.19
Industries	0.40	2.3	0.06	0.10	0.45	Area 3	0.03	1.0	0.01	0.01	0.02
Houses	0.17	1.3	0.03	0.04	0.13	Area 4	0.04	3.1	0.01	0.01	0.02
Apartments	0.40	2.0	0.05	0.10	0.30	Area 5	0.030	2.0	0.01	0.02	0.02
Commercial	0.40	2.3	0.06	0.03	0.30	Total	0.20	2.0	0.03	0.04	0.17
Roads	0.30	2.0	0.10	0.07	0.30						
STFs	0.030	2.0	0.01	0.02	0.02						

studied. However, only residential, commercial and mixed urban land uses were studied. The data were also collected on a large scale, i.e. the data are not representative of individual urban areas where more significant differences may exist. Furthermore, other studies have indicated a large variation of loads coming from different land uses (Novotny, 1992). The standard concentrations, which are land use specific, may especially be used in 'planning-level' analysis and are estimated empirically from a large set of continuous and flow proportional field sampling data, which contributes to their general applicability (Marsalek and Ng, 1989). The NURP study reports a lack of correlation between event runoff volumes (or intensity) and mean concentrations, which makes it acceptable to sample events of any magnitude, without particularly emphasizing large events. This also contributes to the general applicability of using standard concentrations that are extracted from different sampling sites, with different precipitation conditions. According to Schueler (1987) this implies that a single concentration value can be applied for the purpose of estimating pollutant loads.

5.3. Comparisons between standard concentrations and sampled concentrations

An attempt has been made to compare the standard concentrations ($C1$) for Area 1 in Table 3 with the sampled values ($C1^*$). The tot-P concentration has been estimated to be approximately 210 μg/l for Flow 1 (yearly average) from standard values. This concentration is higher than most of the analysed concentration values from the inflow waters (Fig. 7). Most of these values are between 50 and 230 μg/l, i.e. within the same order of magnitude.

The best correlation between sampled and standard concentrations occurs for nitrogen (Flow 1) (see Fig. 7). The corresponding values for tot-N are 1900 μg/l (calculated from standard values) versus 500–2500 (measured). The standard values for the studied metals lead (30 μg/l), copper (50 μg/l) and zinc (180 μg/l) are larger than the sampled lead (2–10 μg/l), copper (5–20μg/l) and zinc (20–110 μg/l) concentrations (see Fig. 8).

5.4. Comparisons with stormwater criteria

The sampled and standard inflow concentrations to the facility have been compared with limit discharge concentrations. These show at which concentrations stormwater may be discharged to recipients without any need of treatment, and have been preliminarily estimated in an adjacent study (Larm, 1998). The limit discharge concentrations are different for different sensitive recipients and can be used for helping to decide when measures are to be taken. The proposed limit discharge concentrations correspond to yearly median values. The limit discharge concentration of tot-P is 125 μg/l and is exceeded by the calculated concentration and is near the median measured concentration (see Fig. 7). The limit concentration of tot-N is 1.7 mg/l and is exceeded by the calculated value of 1.9 mg/l, but not by the median measured concentration. The limit discharge concentrations for Pb (20 μg/l), Cu (25 μg/l) and Zn (175 μg/l) are exceeded by the calculated concentrations, but not by the measured (see Fig. 8).

The limit for eutrophic conditions to prevail has been estimated to be approx. 20–25 μg/l (tot-P) and 400–750 μg/l (tot-N). The limit is exceeded for both phosphorus and nitrogen during every month for Flow 1 and during most months for Flow 2. In comparisons with trophic classifications the measured tot-P concentrations correspond to hypertrophic (very eutrophic/nutrient rich) water conditions (Naturvårdsverket, 1993), especially regarding Flow 1.

5.5. Recipient data

The surface sediment content of both nutrients and metals has decreased during the last 3 years. The content of tot-P has decreased from 1300 to 570 mg/kg DS (dry substance), PO_4-P from 13 to 1.7 mg/kg TS and tot-N from 11 000 to 4300 mg/kg DS. Corresponding metal values are from 31 to 7 for lead, from 96 to 32 for copper and from 470 to 94 mg/kg DS for zinc.

6. Material transport

Mass transport has been quantified from stan-

Table 4
Calculated quantities of land use specific mass flux (left) and calculated loadings from the different inflow areas (right)[a]

Land use	L(P)	L(N)	L(Pb)	L(Cu)	L(Zn)	Area	L(P)	L(N)	L(Pb)	L(Cu)	L(Zn)
Forests	8	309	2.8	2.5	4.6	Area 1	170	1528	28	43	148
Parks	7	644	2	1	2	Area 2	68	651	11	10	51
Industries	45	257	7	11	50	Area 3	2	74	0.7	0.6	1
Houses	63	488	10	16	50	Area 4	1	99	0.4	0.3	0.5
Apartments	61	304	8	15	46	Area 5	1	77	0.3	0.6	0.9
Commercial	42	240	6	3	31	Total	243	2429	41	54	201
Roads	17	112	6	4	17						
STFs	1	77	0.3	0.6	0.9						
Total	243	2429	41	54	201						

[a] Data are given in kg/year.

dard values (calculated flow and standard concentrations). The use of standard concentrations generally provide better estimations compared to quantifications from instantaneously measured concentrations due to the intermittent nature of stormwater. The following formula for quantification of material (mass) transport has been used:

$$L = QC \tag{5}$$

where Q is runoff (m^3/year or m^3/month), L is mass loading rate (mass flux) (kg/year or kg/month) and C is standard concentration (kg/m^3, mg/l or μg/l).

The total pollutant loading of a sub-watershed area is obtained by summation of contributions from individual land use types.

The largest, and about 50%, of the phosphorus load of 240 kg/year on the facilities studied comes from areas with houses and apartments. Other land uses that contribute with P loadings are mainly industries, commercial areas and roads. Only a small part comes from forests, park areas and directly on the STFs from atmospheric dry and wet deposition (Table 4). The nitrogen load mainly comes from areas with parks, houses, forests and apartments. Regarding the studied metals (lead, copper and zinc), these originate to a large extent from areas with houses, apartments and industries.

Areas 1–3 in Table 4 (right part) give rise to point loadings to the facilities, whereas Areas 4 and 5 give rise to diffuse loadings in the form of runoff from surrounding areas to STFs (Area 4) and in the form of direct atmospheric deposition on the STFs (Area 5). An estimation of the share of diffuse loadings to total (point + diffuse) loadings to the facilities in Flemingsbergsviken (Area 4 and 5) shows values between 0.7 and 7.2%: 0.8% (P), 7.2% (N), 1.7% (Pb), 1.7% (Cu) and 0.7% (Zn). These diffuse loadings are often neglected in quantification of the reduction efficiency for STFs and can be much larger than those in this case study, especially if the STF is located further upstream in the watershed.

The largest uncertainties in the material transport calculations concern runoff coefficients and standard concentrations. Other uncertainties concern sampling and analysis procedures. The reduction efficiency and monthly mass loads are not presented since the available flow and concentration data is uncertain. However, the reduction efficiency is discussed below.

7. Discussion

Flow and concentration data, such as those presented in this paper, are needed for estimations of the reduction efficiency of a stormwater treatment facility. However, the reliability of the estimation is dependent on different factors, such as the monitoring programme. Some of the most important factors influencing the estimation of the reduction are discussed here. Generally, the efficiency is expected to be lower during the first few years before soil and vegetation conditions are more stabilised. A monitoring programme should therefore be in operation at least 3–5 years. A continued high efficiency requires the facility to be maintained properly regarding sediment removal and plant harvesting.

Flow measurements can provide uncertain values due to water leakage, back flows from the recipient, low altitude differences and difficulties that are related to wintertime operation (freezing problems). The reduction efficiency is also dependent on inflow concentrations, with generally lower efficiency at low concentrations. In the inflow waters to Flemingsbergsviken, the metal concentrations are low, the phosphorus concentrations are relatively low and the nitrogen concentrations are normal. The low concentrations can partly be explained by drainage water from large green areas being discharged to incoming ditches. It may be discussed if Flow 2 should be conducted to the facility or go directly to the lake recipient, since the nutrient concentrations of this flow are lower than in the outflow water from the facility. However, this may not be the case with metals. There is not enough data as a basis for deciding how to handle Flow 2.

Other uncertainties regarding the estimation of reduction efficiency concern metal retention, sedimentation and plant uptake in the incoming ditches. This means that the reduction values should be higher if these ditches are looked upon as parts of the facility. However, in Flemingsbergsviken there is no sampling further upstream in the ditches (Larm, 1999). Generally, the presented standard concentrations are higher than the sampled concentrations at the inflows to the facility. The correlation would probably have been closer if the reduction in the open ditches had been considered.

A problem with the instantaneous sampling at Flemingsbergsviken, when for example one sample is to represent a whole month, is that the samples were collected during a short period of time, around one hour. However, it takes the water on average 3–5 days to pass through the whole facility. This means that different water is sampled. In principle, the sampling of the inflow water can occur a couple of days after a rain event when the largest pollutant loads (and the largest concentration) already have passed the sampling point. In such a case, the analysis of this sample exhibits low concentration values. When the outflow point is sampled 1 h later, perhaps the water with the higher pollutant concentration has reached this point. The result is a low value of reduction efficiency. Such conditions give rise to increased uncertainty regarding monthly estimated reduction values. The error in a longer time perspective may be less, but if an estimated efficiency for a specific month shows a negative or a low value and the flow is large, then the error may be large even on a yearly basis. Furthermore, some analyses show values below the detection limits, also adding to the uncertainty.

Another factor related to the estimation of reduction efficiency concerns the use of the area before the construction works. The area at Flemingsbergsviken consisted of a straight open ditch quickly transporting the stormwater to the recipient. During high flow periods the surrounding grass area was flooded by polluted stormwater. During a long period of time, nutrients and metals have accumulated in the sediments of this area. When the construction of the facility began in 1995 there was probably large pollutant amounts left in the sediments. These sediments were affected by the construction works. Such disturbed sediments may give rise to leakage of pollutants from the sediments. One effect of this is that estimated reduction efficiencies become lower than the case would have been if this pollutant release would not have been occurring. Furthermore, the method of sampling a predestinated day each month can generally lead to too low concentrations, since often the highest concentrations, occurring during the first period of the rain event (the 'first-flush' effect), are missed.

8. Conclusions

A spreadsheet model which requires relatively little input data can be used for calculating long-term runoff water flow by employing land use specific runoff coefficients, precipitation data and estimated watershed area. Yearly and monthly material transport can be estimated from calculated or measured flow and standard or measured pollutant concentrations. The results from such a model and from measurements show that flow and concentrations vary significantly between different land uses and indicate that the monthly runoff coefficients and the concentrations are the most uncertain parameters in the quantification of material transport. The runoff coefficients are typically largest during the snow melt period. The measured flow data and the instantaneous concentration samples can not be used for calibrating the model, but comparisons with calculated values may nevertheless be interesting. For example, most of the sampled concentrations were lower than the standard concentrations. Explanations could, for instance, be an occurring reduction in the inflow open ditches or the method of instantaneous sampling a predestinated day. The best correlations were for nutrients, especially for nitrogen. The standard concentrations are believed to be more reliable than the instantaneously sampled concentrations.

Comparisons of quality data to preliminary estimated stormwater criteria (limit discharge concentrations) indicate a need of reducing especially the metal loading but also the nutrient loading on the lake recipient. It is impossible to estimate reliable values of reduction efficiency of the facility due to uncertain flow measurements and instantaneously sampled concentrations, as discussed earlier. However, there are indications of decreased concentrations of phosphorus and metals at the outflow. The sediment content of both phosphorus, nitrogen, as well as of metals, in the lake recipient has decreased significantly during the last three years.

The observations from this study indicate that flow should be measured at the major inflows and especially at the outflow of a facility. However, many problems may occur and therefore the equipment for flow measurements ought to be carefully chosen, also considering the type of weirs and suitable sites for measuring. Carefully planned monitoring programmes are required for more reliable estimations of concentrations, flows and reduction efficiency. Such programmes can also be used to calibrate material transport models. Flow proportional sampling using continuous flow indicators and an automatic sampling device seems to be most appropriate. Collected samples may be analysed to decrease the costs. If instantaneous sampling is used all the same, uncertainty analyses are needed for estimation of material transport and reduction efficiency. The harvested plant material is suggested to be analysed regarding its nutrient and metal content. We emphasize that the same water should be sampled in the inflow and the outflow waters of facilities. This suggests the time of sampling the outflow is prolonged to a time that corresponds to the predicted water detention time.

Pilot studies using models, similar to the developed model Stormtac, are useful for establishing a monitoring programme. The models may provide indications of expected pollutant concentrations for avoiding analyses below the detection limits and may be used for comparisons with sampling results. Land use specific standard concentrations are more appropriate for use rather than instantaneous concentration data for the quantification of material transport. The quantification of material loadings is suggested to include both point and diffuse loadings to/from the facilities and seasonal along with shorter time variations. This is especially valid when there is more than one inflow, when surrounding areas (including groundwater and atmosphere) are estimated to contribute with a non-negligible inflow (or infiltration and percolation to the groundwater).

The experiences from the first years of operation of the facility and the monitoring programme have contributed to an increased understanding of problems (for example, concerning flow measurements) and uncertainties that may occur. They may also lead to improved design criteria and to make more effective programmes for monitoring and maintenance.

9. Notation

*	Index for value of sampling or measurement
C	Concentration (mg/l or μg/l)
L	Mass flux (kg/year or kg/month)
p	Precipitation intensity (m/year or m/month)
Q	Runoff water flow (m^3/year or m^3/month)
Q_{dim}	Dimensioning water flow (l/s)

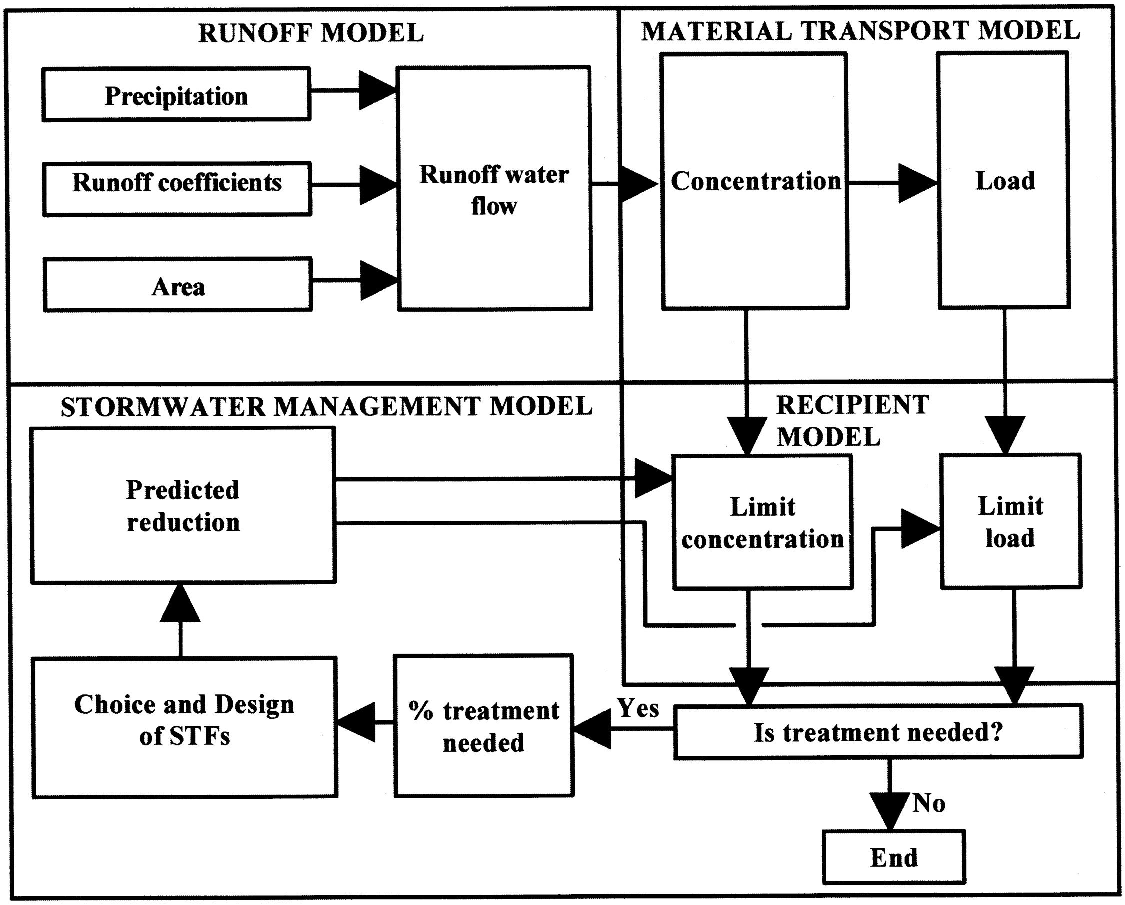


Fig. 9. Flowchart for the watershed management model Stormtac.

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Appendix A. Model description

The model used and developed in this study is the spreadsheet (Excel) model Stormtac (see Fig. 9). Stormtac is a watershed management model for the quantification of monthly and yearly material transport and for the design of stormwater treatment facilities. It identifies the contribution of different land uses. The model equations (of which Eqs. (3)–(5) are related to material transport) are simple, but as such are consistent with available data. The equations consider base flow, and according to Schueler (1987) larger residential watersheds, such as this case study, often generate appreciable volumes of base flow. Furthermore, the method developed here employs different land use specific standard concentrations. Generally, standard values can provide a relatively good description of the pollutant transport over longer time periods (Larm, 1996). The standard concentrations are assumed to be constant at all times for a certain pollutant. By multiplying an annual runoff volume with this concentration, an annual runoff load (Eq. (5)) is produced. However, if a hydrological model that produces variable flows is coupled to these concentrations, the loads will vary (Water Environment Federation and American Society of Civil Engineers, 1998).

Here, a spreadsheet has been developed to automate and extend the standard concentration approach and to calculate the runoff volumes simply from precipitation data, land use specific areas and runoff coefficients (Eq. (3)). According to Water Environment Federation and American Society of Civil Engineers (1998) such a spreadsheet approach is best suited to long-term loads. The advantages are that it easily simulates a mixture of land uses and estimates overall loads and flow-weighted concentrations for the study area. Stormtac is also extended to include estimations of the effectiveness of STFs either by applying constant reduction efficiencies from literature data or using empirically functions between inflow and outflow concentrations. The use of unit loads is an alternative simple concept, consisting of pollutant values of mass per area per time (e.g. expressed in $kg/km^2/year$). However, the unit load is based on an average or 'typical' runoff volume and can not vary from year to year (Water Environment Federation and American Society of Civil Engineers, 1998).

The model also includes several sub-models for the design of wet ponds, wetlands, filter strips and open ditches. Stormtac has been verified and calibrated regarding flow and concentrations to a couple of case studies and is continuously being updated and calibrated with data.

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