The Effect of Wastewater and Treatment System on Effluent Quality

1. Introduction

The design of wastewater treatment systems is generally based on the expected average conditions of wastewater quantity and quality and on the required treatment efficiency. This factor determines the type of treatment system, whereas its performance is either estimated by means of empirical guide values or experimentally evaluated.

The 'guide value' approach is briefly summarized in section 2 with respect to biological systems treating municipal wastewaters. Principal effluent quality goals are the removal of organic matter (BOD) and possibly nitrification of ammonia in order to reduce the load of oxygen consuming matter discharged to the receiving water. This approach is based on average conditions of plant input and plant performance, and gives, therefore, information only on the average effluent quality. It is, however, common experience that effluent quality parameters vary considerably. Possible causes of such variations are:

- variations of the raw wastewater quantity and quality (concentration and kind of organic matter), affecting the reactor retention times and the organic loading rate of the biological treatment system;
- variations of the wastewater temperature, influencing the rates of sedimentation and of biological degradation of organics as well as of nitrification;
- variations of the concentration of the conservative total solids (salts), mainly of industrial origin, affecting the rate of biological degradation via changes in the osmotic pressure;
- variations of miscellaneous factors, such as pH, nutrient concentrations, and inhibitory or even toxic substances.

The above factors and their effects can be designated as the 'inherent' or 'natural' variability of well designed and operated treatment plants; inherent because little, if any, control can be exerted in order to reduce their effect on the effluent quality. At last, not 'inherent' factor causing effluent variability may be added: extremely large variations of the forementioned factors or incorrect control of operation may upset parts of the treatment system. This can significantly contribute to effluent quality variability. The third section of this paper is devoted to the problem of 'variability' and intends:

- to give information on effluent quality variations based on data of numerous municipal and industrial wastewater treatment plants. No attempt is made to correlate the variability with any of the above causative factors; instead a stochastic description of variability is aimed at;
- to develop causal relations between some of the forementioned driving factors of variability and the type of treatment system on the one hand, and the effluent quality variation on the other hand. Only highly simplified models to describe input variation and treatment response will be applied;
- to outline some methods for reducing effluent quality variations.

2. Constant Wastewater Input and Treatment Response

2.1. Wastewater Input

Within the forementioned 'guide value' approach the wastewater input of a treatment system is generally taken to be constant. The input is either measured or based on per-capita-daily amounts of sewage q and pollutants w. Typical values are for sewage q = 0.150 to 0.250 m³/person/day and for the important pollutants BOD, N and P, respectively, w = 10 to 15 g N/person/day and w = 3 to 6 g P/person/day. The total flow Q and waste loads W_q (g/d) are then estimated from the number of population equivalents p by Q = q.p and W_q = p.w_q, whereas the pollutant concentrations c_i (g/m³) are given by c_i = w_i/q.

2.2. Treatment Response

2.2.1. Primary Sedimentation

Most of the settleable solids are removed by primary sedimentation which is designed on the basis of the hydraulic detention time \( \tau \) in the settling tank (e.g. 2 to 4 hours) and the hydraulic surface loading rate (m³ sewage per m² tank surface area per hour). Typical treatment results of primary sedimentation are 30 to 40 % removal of BOD, 2 to 10 % removal of N and P.

The remaining waste loads constitute the input loads F to the biological part of the treatment system.

2.2.2. Biological Treatment

The response of biological treatment systems (activated sludge and trickling filter) is mainly determined by the food-micro-organism ratio \( F/M \), i.e. the waste loading rate \( F \) (g/d) related to the mass of micro-organisms, M, active in degradation of organics. In activated-sludge systems M is generally indicated by the mass of activated sludge suspended solids in the aeration tank:

\[ M = V \cdot G \]

whereof: \( M' = V \cdot G \) (g solids/d) or \( F/M' = F/V \cdot G \) (g BOD/g volatile solids/d) and can be designated as the sludge loading rate of the system. With trickling filters, the mass of micro-organisms is taken as being proportional to the filter volume. Hence \( F/M \) is estimated by the volumetric loading rate with organics: \( F/M'' = F/V \) (g BOD/m² filter volume/d).

Figures 1 and 2 show typical correlations between the F/M ratio (as defined above) and the BOD removal efficiency \( \eta \) (%) for activated sludge and trickling filter.
systems, respectively. With decreasing F/M ratio not only does the efficiency increase, but also the amount of biological sludge produced by the system is reduced owing to increasing endogenous respiration. The reduction of sludge production when decreasing the F/M-ratio of the biological treatment system has implications on the removal of the nutrients N and P.

When assuming assimilation into bio-mass as the only pathway of removal, and a constant composition of the biological sludge, it is obvious that the removal efficiency of both nutrients increases with increasing F/M ratio. Nevertheless, this removal efficiency will always be small and rarely exceed 40 %.

A second implication of the reduced sludge growth at low F/M-ratios is nitrification. Since nitrifying micro-organisms multiply at a much slower rate than heterotrophic bacteria, the former are washed out of the system together with the waste sludge at high rates of excess sludge production, i.e. at higher F/M-ratios.

It is common experience, therefore, that nitrification takes place at fairly low F/M-ratios: within the activated sludge system at F/M < 0,15 to 0,20 g BOD/g solids, d and within trickling filter systems at loading rates below 300 g BOD/m² d. Decreasing the F/M-ratio below the magnitude required for nitrification has two consequences.

— owing to the prevailing endogenous respiration the sludge is fairly well stabilized or mineralized;

— the nitrate, produced by nitrification, is partially denitrified to nitrogen at locations where anaerobic conditions occur within the reactor. This mechanism can increase the efficiency of nitrogen removal considerably.

2.2.3. Secondary Sedimentation

The bio-mass produced by trickling filters and the recirculated and excess sludge of activated-sludge systems is removed from the sewage by secondary sedimentation. Secondary clarifiers for trickling filters are generally designed like primary ones on the basis of retention time and surface loading rate.

The high solids concentration of the aeration tank effluent requires a more thorough approach. Primary factors governing the sedimentation efficiency are the solids surface-loading rate Rₛ (kg solids per m² tank surface area per hour) and/or the solids volume-surface-loading rate Rᵥ (m³ of sludge solids per m² tank surface area per hour). Both parameters are linked by the sludge volume index SVI (m³ sludge volume per kg solids after 30 min, settling):

\[ Rᵥ = Rₛ \cdot SVI \]

Typical values for Rᵥ are 2.0 to 3.5 kg solids/m² . h for an index of SVI = 0.1 m³/kg. At higher sludge volume indices the mass loading rate should be reduced accordingly. The effluent suspended solids concentration is likely to increase significantly when applying higher solids surface loading rates. This, again, will contribute to the total effluent BOD.

2.2.4. Conclusions

Under steady state conditions the effect of wastewaster and treatment system on effluent quality is determined by the average quantity and quality of the wastewater input and the average removal efficiency of the chosen system. The effect of the sedimentation units is primarily given by the selected wastewater retention times, whereas the influence of the biological part of the treatment system on the effluent quality is mainly governed by the chosen loading rate with organics (F/M-ratio). A corresponding differation of the systems is given in table 1.

The removal efficiency of N and P by denitrification of nitrate have additional effects on the effluent quality.

3. Variable Wastewater Input and Treatment Response

3.1. Stochastic Description of Effluent Variability

The variable wastewater input quantity and quality, the variation of wastewater temperature, of salt concentration, pH, nutrient concentration, etc., and also their effect on the response of the treatment system, cause a considerable variability in the effluent quality. If, as a first approach, all these effects are considered to be of random nature, then the variability of effluent quality parameters can be generalized by simple stochastic models, e.g. by a Gaussian normal distribution.

Evaluation of the mean

\[ M = \frac{\Sigma X}{n} \quad (1) \]

and of the standard deviation

\[ s = \sqrt{\frac{\Sigma (X - M)^2}{n - 1}} \quad (2) \]

of all parameters X is all that is required. The relative variability is indicated by the coefficient of variation CV

\[ CV = s/M \quad (3) \]

Effluent parameters, however, frequently show not a symmetrical but rather a positively skewed distribution, the mean being greater than the mode. The skewness can be
TABLE 2 - Characterization of Sewage Treatment Plants (1).

<table>
<thead>
<tr>
<th>Plant</th>
<th>Type</th>
<th>Flow 10^3 m³/d</th>
<th>Dry solids in aeration tank kg/m³</th>
<th>Sludge loading rate kg BOD g/m³ solids d</th>
<th>Volumetric loading rate m³ d</th>
<th>Retention time in secondary clarifier h</th>
<th>Samples</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cleveland Eastery, Ohio, USA</td>
<td>AS</td>
<td>370</td>
<td>1.25</td>
<td>0.24</td>
<td>300</td>
<td>7.5</td>
<td>3.0</td>
</tr>
<tr>
<td>Cranston, R.I., USA</td>
<td>AS</td>
<td>16.4</td>
<td>1.40</td>
<td>0.53</td>
<td>740</td>
<td>5.7</td>
<td>2.0</td>
</tr>
<tr>
<td>Gary, Ind., USA</td>
<td>AS</td>
<td>112</td>
<td>3.30</td>
<td>0.12</td>
<td>400</td>
<td>5.0</td>
<td>2.0</td>
</tr>
<tr>
<td>Heilbronn, Germany</td>
<td>AS</td>
<td>6.9</td>
<td>2.20</td>
<td>0.55</td>
<td>1200</td>
<td>12.0</td>
<td>4.5</td>
</tr>
<tr>
<td>Maple Lodge, England</td>
<td>AS</td>
<td>93</td>
<td>4.60</td>
<td>0.10</td>
<td>460</td>
<td>12.0</td>
<td>4.5</td>
</tr>
<tr>
<td>Marion, Ind., USA</td>
<td>AS</td>
<td>22.5</td>
<td>2.60</td>
<td>0.20</td>
<td>520</td>
<td>4.0</td>
<td>2.0</td>
</tr>
<tr>
<td>Slough, England</td>
<td>TF</td>
<td>17.3</td>
<td>2.40</td>
<td>0.60</td>
<td>1440</td>
<td>7.2</td>
<td>3.5</td>
</tr>
</tbody>
</table>

AS = conventional activated sludge plant
TF = trickling filter plant

TABLE 3 - Frequency Analysis of 'daily data' (1).

<table>
<thead>
<tr>
<th>Plant</th>
<th>Parameter</th>
<th>Dimension</th>
<th>Mean of input</th>
<th>Variation of input</th>
<th>Correct freq. distribution</th>
<th>Samples</th>
</tr>
</thead>
<tbody>
<tr>
<td>Heilbronn</td>
<td>BOD</td>
<td>g/m³</td>
<td>160.6</td>
<td>15.8</td>
<td>43.5</td>
<td>x</td>
</tr>
<tr>
<td>Maple</td>
<td>BOD</td>
<td>g/m³</td>
<td>219.9</td>
<td>16.8</td>
<td>20.4</td>
<td>—</td>
</tr>
<tr>
<td>Lodge</td>
<td>BOD</td>
<td>g/m³</td>
<td>215.4</td>
<td>15.4</td>
<td>1.225</td>
<td>x</td>
</tr>
<tr>
<td></td>
<td></td>
<td>kg/d</td>
<td>1576</td>
<td>39.9</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td></td>
<td>flow m³/d</td>
<td>94262</td>
<td>10.7</td>
<td>1.112</td>
<td>—</td>
</tr>
<tr>
<td>Slough</td>
<td>BOD</td>
<td>g/m³</td>
<td>437.7</td>
<td>20.5</td>
<td>3.30</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td></td>
<td>kg/d</td>
<td>428</td>
<td>1.67</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td></td>
<td>sus. solids</td>
<td>18.4</td>
<td>43.2</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td></td>
<td>NH₄-N</td>
<td>9.4</td>
<td>62.9</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td></td>
<td>NO₃-N</td>
<td>0.82</td>
<td>38.8</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td></td>
<td>NO₂-N</td>
<td>0.76</td>
<td>1.485</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td></td>
<td>flow m³/d</td>
<td>17279</td>
<td>22.6</td>
<td>x(a&gt;0)</td>
<td>—</td>
</tr>
</tbody>
</table>

Explanation:
There are 2 rows of results for each parameter:
— first row: results on the basis of a normal frequency distribution
— second row: results on the basis of a log-normal frequency distribution.
The column 'mean' states the corresponding means.
The column 'variation' states in the first row: the coefficient of variation
— second row: the log-normal standard deviation.
The column 'correct frequency distribution' marks the type of distribution for which skewness and curtosis are not significant; (x) indicates insignificant skewness but significant curtosis.
Significance tests on a 5% level.

eliminated by applying the log-normal distribution with the log-mean
\[ M_l = \exp \left[ \frac{\Sigma \ln X - n \ln \bar{M}_l}{n - 1} \right] \] (4)

and the log standard deviation
\[ s_l = \exp \left( \frac{\left( \Sigma (\ln X - \ln \bar{M}_l)^2 \right)}{n - 1} \right)^{1/2} \] (5)

frequency distribution, therefore, can be based on assuming ln(X + a) to be normally distributed. The constant 'a' can be estimated from the skewness of the data, as has been shown elsewhere (2). The data X of effluent quality parameters have to be obtained by sampling and the question arises what kind of samples should be taken. The possibilities are — grab samples, which include all the variation;
— composite samples, taken over a certain averaging time, e.g. one day, one week, etc.

The variation within the averaging time is excluded by this approach.
Also the data availability determines the selection of the type of data. Composite daily data are frequently available and data for longer averaging times, e.g. 'monthly' data, can easily be computed from the foregoing. Although grab sample data may be available, they generally lack randomness and completeness over a longer period.

Available data from a number of treatment works were evaluated according to the forementioned concept. The treatment plants are characterized in table 2 (most are 'low loaded' systems) and the type and number of samples is indicated:
1. daily composite samples, indicated as 'daily samples', and
2. monthly composite samples, obtained by compiling daily composite samples, indicated as 'monthly samples'.
The number of samples ranges from 72 to 307 for daily samples and from 71 to 193 (6 to 16 years) for monthly samples.

The following statistics were computed for all parameters:
1. mean
2. standard deviation
3. skewness (g₁)
4. curtosis (g₂)
The results are summarized in table 3 (analysis of 'daily data') and table 4 (analysis of 'monthly data') in two rows for each parameter:

— the first row refers to the normal distribution and states the in- and output means and the respective coefficients of variation (in %) to indicate the parameter variability;
— the second row refers to the log-normal distribution and states the geometric means of in- and output and the log-normal standard deviation for characterization of the parameter variability.

The input data refer to the inflow of the biological treatment unit (i.e. effluent from primary sedimentation) the output data to the effluent from the biological unit (i.e. plant effluent).

The last column of tables 3 and 4 marks the correct type of frequency distribution, obtained from a significance test on skewness and curtosis at a 5 % level.
With some of the parameters, the original data had a positive skewness ($g_1 > 0$), which became negative ($g_1 < 0$) by the log-transformation. When curtosis was statistically not significant in both cases a value of $a > 0$ could be found such that $(X + a)$ is log-normally distributed. These parameters are indicated 'X (a > 0)' in the last column.

Two examples are represented by means of a probability plot in figures 3 and 4.

The foregoing information on variability of influent and effluent concentration of Union Carbide plants treating paper mill wastewater (table 7); petrochemical wastewater (after [4]); municipal wastewater at Houston, Texas (table 5); and samples in- and output and the log-normal standard deviation of in- and output concentration to indicate the parameter variability.

There are 2 rows of results for each parameter:
— first row: results on the basis of a normal frequency distribution
— second row: results on the basis of a log-normal frequency distribution.

All daily data are log-normally distributed. $s_i$ is estimated from CV.

The following conclusions may be drawn from the compiled results:

1. Most of the parameters can be generalized by a log-normal frequency distribution. This is also true for the 'monthly data' (table 4).

2. All parameters show considerable variation: the coefficients of variation range from 11 to 164 %, the log-normal standard deviations from 1,11 to 3,13.

3. The effluent variability (CV or $s_i$) of a parameter is always greater than its influent variability (tables 3, 4 and 6). This is to be expected, since the variabilities of influent and treatment response together determine the effluent variation. To make this conclusion clear, it should be stated that the (normal) standard deviation $s$ of a parameter is reduced by treatment. Since, however, the mean $M$ is decreased to a greater extent, an increase of the coefficient of variation $s/M$ is always observed. This is illustrated by the BOD-concentration of the

### Table 3 - Frequency Analysis of 'daily data' (1).

<table>
<thead>
<tr>
<th>Plant</th>
<th>Parameter</th>
<th>Dimension</th>
<th>Mean of plant input</th>
<th>Variation of plant input</th>
<th>Correct freq. distribution input</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cleveland</td>
<td>BOD</td>
<td>g/m³</td>
<td>95,6</td>
<td>11,5</td>
<td>25,2</td>
</tr>
<tr>
<td>Cranston</td>
<td>BOD</td>
<td>g/m³</td>
<td>92,6</td>
<td>10,6</td>
<td>1,290</td>
</tr>
<tr>
<td>Gary</td>
<td>BOD</td>
<td>g/m³</td>
<td>168,1</td>
<td>23,3</td>
<td>1,370</td>
</tr>
<tr>
<td>Maple</td>
<td>BOD</td>
<td>g/m³</td>
<td>84,4</td>
<td>12,8</td>
<td>25,6</td>
</tr>
<tr>
<td>Lodge</td>
<td>BOD</td>
<td>kg/d</td>
<td>31890</td>
<td>1330</td>
<td>36,2</td>
</tr>
<tr>
<td></td>
<td>flow</td>
<td>m³/d</td>
<td>85750</td>
<td>31,0</td>
<td></td>
</tr>
</tbody>
</table>

| Marion   | BOD       | g/m³      | 85,1                | 8,4                      | 33,7                            |
| Slough   | BOD       | g/m³      | 81,4                | 7,5                      | 23,9                            |
| BOD      | kg/d      | 503,4     | 23,0                | 1,154                    | 1,232                           |
|          | flow      | m³/d      | 15840               | 742                      | 15,7                            |
|          |           |           | 15640               | 715                      |                                  |
|          |           |           | 81350               | 1,400                    |                                  |

### Table 4 - Frequency Analysis of 'monthly data' (1).

<table>
<thead>
<tr>
<th>Plant</th>
<th>Parameter</th>
<th>Dimension</th>
<th>Mean of plant output</th>
<th>Variation of plant output</th>
<th>Correct freq. distribution output</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cleveland</td>
<td>BOD</td>
<td>g/m³</td>
<td>176,7</td>
<td>26,9</td>
<td>32,2</td>
</tr>
<tr>
<td>Cranston</td>
<td>BOD</td>
<td>g/m³</td>
<td>168,1</td>
<td>23,3</td>
<td>1,370</td>
</tr>
<tr>
<td>Gary</td>
<td>BOD</td>
<td>g/m³</td>
<td>84,4</td>
<td>12,8</td>
<td>25,6</td>
</tr>
<tr>
<td>Maple</td>
<td>BOD</td>
<td>g/m³</td>
<td>370,2</td>
<td>16,2</td>
<td>16,4</td>
</tr>
<tr>
<td>Lodge</td>
<td>BOD</td>
<td>kg/d</td>
<td>31890</td>
<td>1330</td>
<td>36,2</td>
</tr>
<tr>
<td></td>
<td>flow</td>
<td>m³/d</td>
<td>29720</td>
<td>1206</td>
<td>1,472</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>85750</td>
<td>31,0</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>81350</td>
<td>1,400</td>
<td></td>
</tr>
<tr>
<td>Marion</td>
<td>BOD</td>
<td>g/m³</td>
<td>85,1</td>
<td>8,4</td>
<td>33,7</td>
</tr>
<tr>
<td>Slough</td>
<td>BOD</td>
<td>g/m³</td>
<td>81,4</td>
<td>7,5</td>
<td>23,9</td>
</tr>
<tr>
<td>BOD</td>
<td>kg/d</td>
<td>503,4</td>
<td>23,0</td>
<td>1,154</td>
<td>1,232</td>
</tr>
<tr>
<td></td>
<td>flow</td>
<td>m³/d</td>
<td>15840</td>
<td>742</td>
<td>15,7</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>15640</td>
<td>715</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>81350</td>
<td>1,400</td>
<td></td>
</tr>
</tbody>
</table>

### Table 5 - Variability of effluent suspended solids concentration (g/m³) at 3 activated sludge systems in Houston, Texas (after [4]).

<table>
<thead>
<tr>
<th>Plant and data</th>
<th>$n$</th>
<th>coefficient of variation CV (%)</th>
<th>log-normal standard deviation $s_i$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Southwest daily</td>
<td>287</td>
<td>84</td>
<td>2,07</td>
</tr>
<tr>
<td>Northeast daily</td>
<td>100</td>
<td>152</td>
<td>2,99</td>
</tr>
<tr>
<td>Slough daily</td>
<td>23</td>
<td>45</td>
<td>1,54</td>
</tr>
<tr>
<td>Sagemont daily</td>
<td>10</td>
<td>79</td>
<td>2,00</td>
</tr>
<tr>
<td>Monthly</td>
<td>12</td>
<td>45</td>
<td>1,50</td>
</tr>
</tbody>
</table>

All daily data are log-normally distributed. $s_i$ is estimated from CV.

### Table 6 - Variability of influent and effluent concentration of 3 Union Carbide plants treating petrochemical waste (after [4]).

<table>
<thead>
<tr>
<th>Plant and samples</th>
<th>influent</th>
<th>effluent</th>
</tr>
</thead>
<tbody>
<tr>
<td>Complete mix activated sludge, F/M ≈ 0,1; $r$ ≈ 1 day</td>
<td>CV = 38 %</td>
<td>CV = 44 %</td>
</tr>
<tr>
<td>weekly composite soluble COD samples</td>
<td>$s_i = 1,44$</td>
<td>$s_i = 2,08$</td>
</tr>
<tr>
<td>Complete mix activated sludge, F/M ≈ 0,2; $r$ ≈ 2,5 days</td>
<td>CV = 33 %</td>
<td>CV = 65 %</td>
</tr>
<tr>
<td>daily composite BOD samples</td>
<td>$s_i = 1,38$</td>
<td>$s_i = 1,81$</td>
</tr>
<tr>
<td>Facultative ponds in series</td>
<td>CV = 31 %</td>
<td></td>
</tr>
<tr>
<td>$r$ ≈ 20 days, 3 day composite BOD samples</td>
<td>$s_i = 1,35$</td>
<td>$s_i = 1,35$</td>
</tr>
</tbody>
</table>

$s_i$ is estimated from CV.
The coefficient of variation is increased from 20.4 to 39.1%, the log-normal standard deviation from 1.225 to 1.468, whereas the standard deviation is reduced from $s = CV \cdot M = 0.204 \cdot 219.9 = 44.9$ to 0.391. 16.8 = 6.57 g BOD/m³, a reduction of 85%. Thus the absolute magnitude of the variations is reduced by treatment, the relative variability ($CV, s$) is increased.

4. With regard to the magnitude of the relative variability of effluent parameters, it is found by comparison of the respective coefficients of variation and log-normal standard deviations that

- the wastewater flow shows the smallest variability;
- the effluent concentrations have a medium variability;
- the effluent loads (kg/d) are of greatest variability.

Again, this is to be expected, since the load variation is made up of flow and concentration variability.

The 'monthly data' show a much greater variability than would be expected from the variation of the respective daily data (compare tables 3 and 4, see table 5). If the data were randomly distributed over the sampling period, then the standard deviation of the monthly data $s_m$ could be estimated from that of the daily data $s_d$ based on the number of samples taken per month $n_m$:

$$s_m = s_d / \sqrt{n_m} \quad (6)$$

or

$$s_m = s_d \sqrt{n_m} \quad (7)$$

From $s_d = 1.468$ for the daily effluent BOD-concentration of the Maple Lodge plant (table 3) and an average number of samples per month of $245/12 = 20.4$ (table 2), one would expect a log-normal standard deviation for the monthly data of $s_{m} = 1.468 / \sqrt{20.4} = 1.089$, whereas the actual value (table 4) is 1.536.

The significant differences between 'expected' and actual values of the CV and $s$ of the monthly data are caused
- by the non-randomness of the data within the sampling period;
- by possible trends of the data within the sampling period.

Both factors bias the true stochastic variability and lead to an overestimate thereof.

The non-randomness may have various causes:
- season dependent water consumption and rainfall (combined sewerage system) give rise to season dependent wastewater flows and hence reactor retention times and treatment efficiencies;
- season dependent wastewater temperatures affecting season dependent treatment efficiencies;
- season dependent wastewater transport times affecting season dependent treatment efficiencies.

Maple Lodge works (table 3): The coefficient of variation is increased from 20.4 to 39.1%, the log-normal standard deviation from 1.225 to 1.468, whereas the standard deviation is reduced from $s = CV \cdot M = 0.204 \cdot 219.9 = 44.9$ to 0.391. 16.8 = 6.57 g BOD/m³, a reduction of 85%. Thus the absolute magnitude of the variations is reduced by treatment, the relative variability ($CV, s$) is increased.

4. With regard to the magnitude of the relative variability of effluent parameters, it is found by comparison of the respective coefficients of variation and log-normal standard deviations that

- the wastewater flow shows the smallest variability;
- the effluent concentrations have a medium variability;
- the effluent loads (kg/d) are of greatest variability.

Again, this is to be expected, since the load variation is made up of flow and concentration variability.

The 'monthly data' show a much greater variability than would be expected from the variation of the respective daily data (compare tables 3 and 4, see table 5). If the data were randomly distributed over the sampling period, then the standard deviation of the monthly data $s_m$ could be estimated from that of the daily data $s_d$ based on the number of samples taken per month $n_m$:

$$s_m = s_d / \sqrt{n_m} \quad (6)$$

or

$$s_m = s_d \sqrt{n_m} \quad (7)$$

From $s_d = 1.468$ for the daily effluent BOD-concentration of the Maple Lodge plant (table 3) and an average number of samples per month of $245/12 = 20.4$ (table 2), one would expect a log-normal standard deviation for the monthly data of $s_{m} = 1.468 / \sqrt{20.4} = 1.089$, whereas the actual value (table 4) is 1.536.

The significant differences between 'expected' and actual values of the CV and $s$ of the monthly data are caused
- by the non-randomness of the data within the sampling period;
- by possible trends of the data within the sampling period.

Both factors bias the true stochastic variability and lead to an overestimate thereof.

The non-randomness may have various causes:
- season dependent water consumption and rainfall (combined sewerage system) give rise to season dependent wastewater flows and hence reactor retention times and treatment efficiencies;
- season dependent wastewater temperatures affecting season dependent treatment efficiencies;
- season dependent waste production, brought about by the pattern of life of the
The effect of such causes can be evaluated by computing means and standard deviations for each of the 12 month separately.

As an example, the monthly means and standard deviations of the daily data of the BOD of the Maple Lodge effluent are given in figure 5 (shaded area is ± one standard deviation), whereas figure 6 shows the same evaluation with respect to the monthly data. The trend of long term data can be evaluated, for instance by means of a linear trend model, and the trend variance can be deducted from the total data variance to give an unbiased estimate of the stochastic variance. Trends are caused by an increase in water consumption, waste production, and catchment area of the works (generally positive trend) as well as by an increase in works capacity. Again, the example Maple Lodge serves as an illustration (monthly data over 16 years). Assuming a linear trend model, the following yearly changes can be noticed (trend correlation coefficients in brackets):

1. yearly change of incoming daily BOD load: + 2370 kg/d (0,951);
2. yearly change of incoming BOD concentration: + 4.8 g/m³ (0.367);
3. yearly change of daily sewage flow: + 5300 m³/d (0.929);
4. yearly change of daily effluent BOD load: + 54 kg/d (0.405);
5. yearly change of effluent BOD concentration: - 0.3 g/m³ (-0.185).

3.2. A Simple Modelling Approach

In the foregoing section, all factors that influence the variability of effluent quality have been assumed to be of random nature. Now an attempt is made to treat two significant factors deterministically by simplified modeling, viz.:

- the influence of the variation of the waste load (kg/d) on the effluent concentration;
- the influence of temperature on the treatment efficiency and hence on the effluent quality.

The goal is to evaluate how treatment systems should be designed to reduce effluent quality variation and to what extent such reduction can be accomplished. Basic assumptions within this simplified approach are

- the wastewater flow Q is assumed constant; concentration variations are caused by waste load variations, only;
- first order rate of degradation of BOD: dc/dt = - k \cdot c;
- two extreme flow-through patterns of the biological part of the system: plug flow and complete mixing flow.

Under the above assumption, the average BOD removal efficiency \eta is obtained for plug flow:

<table>
<thead>
<tr>
<th>nr.</th>
<th>Product</th>
<th>Treatment</th>
<th>( r(d) )</th>
<th>Parameter</th>
<th>( n )</th>
<th>CV (%)</th>
<th>( s_i )</th>
<th>type of distribution</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Groundwood</td>
<td>AST</td>
<td>BOD</td>
<td>179</td>
<td>67</td>
<td>1.84</td>
<td>log-normal</td>
<td></td>
</tr>
<tr>
<td>2</td>
<td>Sulfite</td>
<td>ASB 11</td>
<td>BOD</td>
<td>326</td>
<td>35</td>
<td>1.40</td>
<td>log-normal</td>
<td></td>
</tr>
<tr>
<td>3</td>
<td>Sulfite</td>
<td>ASB 9</td>
<td>BOD</td>
<td>333</td>
<td>26</td>
<td>2.89</td>
<td>normal</td>
<td></td>
</tr>
<tr>
<td>4</td>
<td>Bleached kraft</td>
<td>ASB + PS</td>
<td>17 + 7</td>
<td>BOD</td>
<td>296</td>
<td>75</td>
<td>2.21</td>
<td>log-normal</td>
</tr>
<tr>
<td>5</td>
<td>Bleached kraft</td>
<td>ASB + PS</td>
<td>2 + 12</td>
<td>BOD</td>
<td>227</td>
<td>35</td>
<td>1.60</td>
<td>log-normal</td>
</tr>
<tr>
<td>6</td>
<td>Bleached kraft</td>
<td>ASB + PS</td>
<td>14 + 60</td>
<td>BOD</td>
<td>253</td>
<td>30</td>
<td>2.00</td>
<td>log-normal</td>
</tr>
<tr>
<td>7</td>
<td>Bleached kraft</td>
<td>AST + PS</td>
<td>BOD</td>
<td>253</td>
<td>55</td>
<td>1.67</td>
<td>log-normal</td>
<td></td>
</tr>
<tr>
<td>8</td>
<td>Bleached kraft</td>
<td>ASB 23</td>
<td>BOD</td>
<td>216</td>
<td>37</td>
<td>1.43</td>
<td>normal</td>
<td></td>
</tr>
<tr>
<td>9</td>
<td>Bleached kraft</td>
<td>AST</td>
<td>BOD</td>
<td>214</td>
<td>43</td>
<td>1.51</td>
<td>normal</td>
<td></td>
</tr>
<tr>
<td>10</td>
<td>Bleached kraft</td>
<td>AST</td>
<td>BOD</td>
<td>236</td>
<td>56</td>
<td>1.55</td>
<td>log-normal</td>
<td></td>
</tr>
<tr>
<td>11</td>
<td>Deinked</td>
<td>ASB 7</td>
<td>BOD</td>
<td>236</td>
<td>57</td>
<td>1.64</td>
<td>normal</td>
<td></td>
</tr>
</tbody>
</table>

SS: suspended solids, PS: post storage, AST: activated sludge treatment, ASB: aerated stabilization basin treatment

- The trend of long term data can be noticed (trend correlation coefficients in brackets):
- the influence of the variation of the waste load (kg/d) on the effluent concentration;
- the influence of temperature on the treatment efficiency and hence on the effluent quality.

The goal is to evaluate how treatment systems should be designed to reduce effluent quality variation and to what extent such reduction can be accomplished. Basic assumptions within this simplified approach are

- the wastewater flow Q is assumed constant; concentration variations are caused by waste load variations, only;
- first order rate of degradation of BOD: dc/dt = - k \cdot c;
- two extreme flow-through patterns of the biological part of the system: plug flow and complete mixing flow.

Under the above assumption, the average BOD removal efficiency \eta is obtained for plug flow:
\[ \eta_{PF} = 1 - e^{-k\tau} \]  

(8)

complete mixing flow:

\[ \eta_{CM} = \frac{k\tau}{1 + k\tau} \]  

(9)

3.2.1. Effect of Waste Load Variability

In order to evaluate the effect of waste load variation on the effluent quality variability a constant first order decay coefficient \( k \) is presumed.

The waste load variation on the biological part of the system is modelled by a periodic input function (see figure): \( F = F_c + F_p \cdot \sin(\omega t) \)  

(10)

where

- \( F_c = \) constant waste load input (kg/d)
- \( F_p = \) amplitude of periodic waste load input (kg/d)
- \( t = \) time (days)
- \( \omega = \) frequency of input = \( 2\pi / \tau_p \) with \( \tau_p = \) period of input (days)

Typical input periods for municipal wastewater are:

- \( \tau_p = 1 \) day to describe the hourly input variation
- \( \tau_p = 7 \) days to generalize possible weekday variations
- \( \tau_p = 365 \) days for seasonal variations

The effluent concentration is then characterized by eq. 11 and 12, which consist also of a constant term (reflecting the effect of the constant input share) and a periodic term (stating the system response to the periodic input share):

a. plug flow:

\[ c = \frac{F_c}{Q} - \frac{F_p}{Q} \cdot e^{-k\tau} + A_{PF} \cdot \frac{F_p}{Q} \cdot \sin(\omega t - \theta_{PF}) \]  

(11a)

with

- \( A_{PF} = \) reduction factor of input amplitude
- \( \theta_{PF} = \) phase shift

\[ A_{PF} = e^{-k\tau} = 1 - \eta_{PF} \]  

(11b)

\[ \theta_{PF} = \omega \cdot \tau \]  

(11c)

b. complete mixing flow:

\[ c = \frac{F_c}{Q} + \frac{F_p}{Q} \cdot \left( 1 + \frac{k\tau}{1 + k\tau} \right) \cdot \sin(\omega t - \theta_{CM}) \]  

(12a)

with

- \( A_{CM} = \) reduction factor of input amplitude
- \( \theta_{CM} = \) phase shift

\[ A_{CM} = \frac{(1 + k\tau)^2 + (\omega t)^2)^{-1/2}}{1} \]  

(12b)

\[ \theta_{CM} = \arctg \frac{\omega t}{1 + k\tau} \]  

(12c)

From the above equations it is obvious, that in plug flow systems (trickling filter, very long activated sludge tanks) the amplitude of a periodic input is reduced by the average treatment efficiency: in a system having 90% BOD removal efficiency, also the periodic amplitude would be reduced by 90%. Since \( A_{PF} \) is only determined by \( \eta_{PF} \), an increase of the retention time, i.e. of the tank volume (presumably coupled with a decrease in the mixed liquor suspended solids) would have no effect on the amplitude reduction. Only the phase shift would grow, because \( \theta_{PF} \) in terms of time is equal to the retention time \( \tau \).

The response of a complete mixing system (rectangular or round activated sludge tank, oxidation ditches) to a periodic input is different; the amplitude is reduced by degradation (as before) and dilution or equalization. The first and second terms of equation 12b quantify these mechanisms, respectively.

Equation 12b can also be applied to predict the effect of equalization basins on a periodic input. Since it may be assumed that degradation effects are negligible in

![Schematic representation of a periodic input and output.](image-url)
such basins, k and/or $\eta_{OM}$ are zero, hence

$$A'_{OM} = [1 + (\omega \tau)^2]^{-1/2} \quad (13a)$$

$$\theta'_{OM} = \arctg (\omega \tau) \quad (13b)$$

To illustrate the effect of complete mixed activated-sludge systems and equalization basins on the amplitude of a periodic concentration input, the factor $A'_{OM}$ is computed as a function of the input frequency $\omega$ for three hypothetical systems:

a. retention time $\tau = 0.25$ days, $\eta = 85 \%$, a medium loaded plant;
b. retention time $\tau = 0.5$ days, $\eta = 90 \%$, a low loaded system;c. retention time $\tau = 2$ days, $\eta = 95 \%$, an extended aeration system with sludge stabilization.

The results are summarized in figure 7 by plotting $\log A'_{OM}$ versus $\log (1/\tau)$ for the above 3 systems. Furthermore, figure 7 contains the results obtained for $\eta = 0$, i.e. for applying the systems as equalization basins. The ordinate difference between the upper and lower curve of each system indicates the influence of biological degradation on the amplitude reduction $A'_{OM}$, the difference between $A'_{OM} = 1$ and the upper curve shows the effect of equalization and dilution. The information contained in figure 7 is generalized in figure 8 by plotting $\log A'_{OM}$ versus $\log (1/\tau)$, i.e. versus the logarithm of the ratio of retention time $\tau$ over periodic time $\tau_p$. Again equalization and decay effects can be differentiated.

It is clear from figures 7 and 8 that equalization effects prevail at high frequency ($\tau_p \tau$), $\tau_p \tau < 0.05$ or $\tau_p \tau > 20 \tau_p \tau$, then equalization effects are practically absent and degradation only causes a reduction of the input amplitude. Then complete mixing and plug flow systems are equally effective, the amplitude reduction factor amounting to $A = 1 - \eta$ for both systems (eq. 11b and 12b).

In concluding it may be stated, therefore, that the reduction factor of the amplitude of a periodic input to a biological system is at the most $1 - \eta$ (or smaller), whereas equalization basins are effective only if the retention time $\tau$ is significantly longer than 0.05 times the input period time $\tau_p$.

The periodic input, modelled before, may be superimposed by other input functions like:

- a step function, i.e. a sudden change of the input load towards a higher or lower average (e.g. when seasonal industries start or end operation);
- a stochastic input, defined by the standard deviation of the input.

The last input type is now discussed. As shown in section 3.1 the input standard deviation $s_i$ is reduced by the treatment system to the smaller effluent standard deviation $s_{e_i}$. Since the corresponding means are decreased more significantly, the coefficient of variation is increased by treatment. According to Novotny (6), the effluent-influent standard deviation ratio $s_{e_i}/s_{i}$ of a complete mixing system is given by

$$s_{e_i} = \sqrt{\frac{s_i}{\tau}} \cdot \frac{\tau_s}{2 \cdot \tau_s \cdot (1 - k_r)} \quad (14)$$

where $\tau_s$ averaging period of sampling, e.g. one day for 'daily samples'.

It is seen from eq. 14 that the reduction of random variability of the influent concentration is decreased by decay (factor: $1 - \eta$) and by dilution and equalization ($\tau$). The model (eq.14) has been experimentally verified under controlled laboratory conditions (7). Its application to the data of Maple Lodge and Slough (table 3) results in a failure (Maple Lodge: predicted ratio: 0.28; data ratio: 0.15 — Slough: predicted: 0.57; data: 0.25), probably because the data standard deviations include periodic, trend, pulse, and step effects and are not
purely random. It can be shown by means of equation 14 that the coefficient of variation increases by treatment. The effluent-influent CV-ratio is defined by

\[ CV_{fi} = \frac{s_{M_{1}}}{s_{M_{1}}} = \frac{1}{1-\tau_{a}} \cdot \sqrt{1-\tau_{a}} \cdot (1-\eta) \]  

(15)

The above ratio is always greater than one, except for \( \tau_{a} < 2 \cdot \tau \cdot (1-\eta) \). At very short averaging periods of sampling (\( \tau_{a} < 2 \cdot \tau \cdot (1-\eta) \)), however, not the random effects but rather periodic effects are evaluated.

The foregoing modelling approaches lack reality to a certain degree because they are based on:

- a constant wastewater flow
- a hypothetical flow-through pattern
- a first order decay model
- a constant reaction coefficient \( k \)

The most severe limitation is caused by the last assumption. \( k \) is influenced by a number of factors which are not constant in sewage treatment plant operation.

The most important are:

- variations in the type and concentration of the incoming organic matter; the microorganisms have continuously to adapt themselves to the changing substrate environment;
- variations in the concentration of activated sludge suspended solids are not anticipated by the models;
- variations in the wastewater temperature have a significant impact on the reaction coefficient \( k \).

The first effect is illustrated by a very interesting experiment by Conway et al (4). Petrochemical wastewater was treated on a laboratory scale by means of 2 complete mixing activated-sludge systems (\( F/M = 0.7 \text{ d}^{-1} \); \( G_{v} = 2200 \text{ gBOD/m}^{3} \cdot \text{d} \), \( \tau = 0.88 \text{ d} \)). The average input concentration of both plants was 1400 g BOD/m³. One plant, however, received a fairly constant input concentration (obtained by wastewater storage) whereas the second was fed with a highly variable input concentration, being changed every 12 hours.

The results are summarized in table 8.

The effect of feed variation on the effectiveness of the biological system is striking. Compared to the more "constant feed" system, the BOD-removal efficiency is
reduced from 98 to 87%, the first order reaction coefficient $k$, i.e. the activity of the activated sludge, is reduced from 70.9 to 7.69 (1/d), i.e. by as much as 89%.

Similar effects are of great significance also in municipal wastewater treatment systems, although the variability of the input concentration is expected to be less than in the above example (compare CV = 60.7% with the data of table 3). Hence, the previously modelled effects of input variation on effluent quality variability will always be magnified by changes in $k$ (or of the activity) in the biological system which are caused by the influent variation.

3.2.2. Effect of Wastewater Temperature Variation

The wastewater temperature $T$ has significant influence on the reaction coefficient $k$, which is commonly expressed by

$$k_T = k_R \cdot f(\Theta)$$

$$= k_R \cdot f(\Delta T)$$

where

$k_T$ = reaction coefficient at temperature $T$
$k_R$ = reaction coefficient at reference temperature $T_R$  
$\Theta$ = temperature coefficient
$f(\Theta) = \Theta / T - T_R$  

Temperature coefficients for the trickling-filter system range from 1,02 to 1,04 and for the activated-sludge system from 1,00 to 1,03. Combining eq. 16 with eq. 8 and 9, respectively, yields a relation between the removal efficiencies $\eta_T$ at temperature $T$ and $\eta_R$ at the reference temperature $T_R$ for plug flow systems

$$\eta_T = 1 - (1 - \eta_R) f(\Theta)$$  (17)

and complete mixing systems

$$\eta_T = \frac{\eta_R \cdot f(\Theta)}{1 - \eta_R \cdot [1 - f(\Theta)]}$$  (18)

A graphical representation of eq. 16 to 18 is given in figures 9 and 10. From figure 9, $f(\Theta) = \Theta \Delta T$ can be read as a function of $\Delta T$. Within the normal wastewater temperature variation and the cited range of $\Theta$, $f(\Theta)$ is expected to extend from 0.5 to 1.5. The effect of $f(\Theta)$ and the average treatment efficiency at the reference temperature $T_R$ (= average yearly temperature) on the efficiency $\eta_T$ at temperatures other than $T_R$ is depicted in figure 10 for plug flow systems (eq. 17) and complete mixing flow systems (eq. 18). This figure contains also information on the variation in the treatment efficiency of a

— trickling-filter system (TF), based on $\eta_R = 0.8, \Theta = 1.030$ and plug flow conditions;

— activated-sludge system (AS), based on $\eta_R = 0.9, \Theta = 1.015$ and plug flow conditions (outer arrows) and complete mixing conditions (inner arrows) assuming a yearly deviation from the average temperature ($T_B$) of $\Delta T = -7.5^\circ C$ (winter) and $+7.5^\circ C$ (summer).

The following conclusions may be drawn from figure 10. The variation in the treatment efficiency (and hence effluent quality) caused by wastewater temperature changes is not only dependent on the temperature coefficient $\Theta$ and the temperature change $\Delta T$, but also on

— the average treatment efficiency $\eta_R$: highly efficient (low loaded) systems are less susceptible to temperature effects than systems of low efficiency (high load).

At extremely high efficiencies the temperature effects are reduced to almost zero.

— the pattern of flow through the system: the treatment efficiency of plug flow systems is much more sensitive to temperature changes than the performance of complete mixing systems.
— for a given temperature variation (e.g., $T = \pm 7.5 \, ^\circ C$, total temperature range: hence $15 \, ^\circ C$) the average treatment efficiency $\eta_{av}$ of a system has a much greater influence on its temperature sensitivity than its temperature coefficient $\theta$. This is seen from a comparison of the large range of the trickling filter ($\eta_{av} = 0.5; \theta = 1.030$) and the relatively small range of the activated-sludge system ($\eta_{av} = 0.9; \theta = 1.015$) in figure 10.

The last mentioned effect is clearly illustrated by data [8] from the trickling-filter plant at Utrecht (fig. 11) and from the activated-sludge plant at Amsterdam (fig. 12). When comparing extreme values of $\eta_{av}$ and $T$ of both works during summer 1971 and winter 1971/72 and assuming plug flow conditions for both systems, the computed (eq. 16 and 17) temperature coefficients $\theta$ are for the trickling filter 1.023 and for activated-sludge system 1.017 (under complete mixing conditions: 1.052 and 1.060, respectively!). It is obvious, that the different temperature sentivity of both systems is due to their difference in average treatment efficiency and not (as commonly assumed) to different temperature coefficients $\theta$.

3.3. Conclusions: How to Reduce Effluent Quality Variability

In the foregoing sections the variability of wastewater treatment system parameters has been described by stochastic modelling and by simplified deterministic models. In concluding, from these considerations some methods for minimizing the effluent quality variations are discussed. The goal of such attempts may be to reduce the variation of effluent load on receiving waters and thereby the quality variation of rivers and streams. Moreover, the chance of complying with effluent standards, generally conceptualized as fixed effluent concentration (or load) values not to be exceeded, will significantly increase.

Methods for reducing effluent variability can be subdivided into design and operation strategies. With respect to the design of biological treatment systems a high efficiency complete mixing system with long wastewater retention time (e.g. extended aeration) will produce the least variable effluent quality, because

— the amplitude of periodic influent concentrations with moderate and high frequency is reduced by decay and equalization, whereas only decay mechanisms are effective in plug flow systems;

— the random variation of influent concentration is effectively decreased by effects of degradation and dilution;

— all factors that change the magnitude of the reaction coefficient $k$ (or the activity of the microorganisms in degrading organic matter) have a less pronounced effect on the treatment efficiency and hence effluent quality in complete mixing systems as opposed to plug flow systems. Important factors of this kind are

— the wastewater temperature;

— the change of the influent waste concentration and composition on the biological activity ($k$) of the system brought about by continuously forcing the biomass to adapt to the changing substrate environment;

— the mixed liquor suspended solids concentration of an aeration tank ($k$ is proportional to $G$).

These effects decrease as the average treatment efficiency of the system increases.

The only conceivable advantage of plug flow systems is the greater average treatment efficiency under steady state conditions of operation.

Equalization basins (completely mixed) prior to biological treatment are effective only for reducing periodic influent concentrations of high frequency ($\omega < 0.1 \cdot 2\pi$), and their most important function can be seen only in decreasing the effect of qualitative and quantitative waste variability on the reaction coefficient $k$.

Operational strategies for reducing effluent quality variation are most effective when based on the 'feed-forward' principle, i.e. the incoming wastewater flow rate and concentration (e.g. COD, TOD, TOC) are applied to exercise control measures. Possible measures are

— adjustment of the return sludge flow to keep the $F/M$-ratio of an activated-sludge system as constant as possible. The feasibility of this approach is limited by the settling rate and the stored mass of activated-sludge solids. The latter can be adjusted by controlling the excess sludge flow;

— addition of coagulants to the raw wastewater at high concentration of organics in order to equalize the load on the biological system;

— diversion of excess flows and excess wastewater concentration into separate holding tanks and subsequent treatment during periods of low flow (e.g. night);

— bringing into operation of stand-by units.

To illustrate the merits of 'operational strategies' in the practice of wastewater treatment, the effect of a number of the aforementioned measures is shown in figure 13. The effluent quality variation (BOD and suspended-solids concentration) of the 7500 m$^3$/d contact-stabilization plant at Hillsboro, Oregon, USA, has been significantly reduced by the 'new operating strategy' [9]. The strategy is based on monitoring the influent TOC, the activated-sludge solids and dissolved oxygen concentration in the aeration tank, the respiration rate of the aeration tank effluent, and the sludge volume index. By adjusting the return and excess sludge flows, the air flow, and the sludge conditioning and sorption time by means of an experimentally evaluated strategy, an effluent quality complying with the set standard could be obtained.

References


**Notation**

- A: amplitude reduction factor: effluent/influent amplitude of a periodic function
- CM: parameter based on complete mixing flow conditions
- CV: coefficient of variation
- F: load on a biological treatment system (BOD, N, P)
- f(ϕ): phase shift
- G: activated-sludge suspended-solids concentration
- G_v: volatiles of G
- k: first-order reaction coefficient
- k_r: k at reference temperature T
- l: parameter based on log-normal distribution
- M: biological mass active in degrading organics
- M: number of samples
- p: number of population equivalents
- PF: parameter based on plug flow conditions
- Q: wastewater flow rate
- q: specific wastewater flow rate: m³ per person daily
- R_s: solids surface-loading rate
- R_v: solids volume surface-loading rate
- s: variable standard deviation
- SVI: sludge volume index
- T: temperature
- T_R: reference temperature
- W: raw waste load (BOD, N, P)
- w: specific raw waste load in g per person daily
- V: reactor tank volume
- X: quality parameter
- Δ T: temperature difference: T - T_R
- η_p: (BOD) removal efficiency
- η_r: % at reference (average) temperature
- η_T: % at temperature T
- Θ: temperature coefficient
- δ: phase shift
- τ: reactor retention time = V/Q
- t_p: time period of a periodic function
- t_s: averaging time of sampling
- ω: frequency of a periodic function ω = 2 π / t_p

---

**Kankerepidemie in VS**, aldus milieugroep

Het fonds voor de milieubescherming in de VS heeft bij het Amerikaanse Hof van Beroep een aanklacht ingediend waarin van de regering strengere maatstaven voor leidingwater worden geëist, waardoor ook afvalstoffen die vermeedelijk kanker verweken moeten worden verwijderd.

Het fonds zegt dat de maatstaven die zijn voorgesteld door het Amerikaanse bureau voor bescherming van het milieu (EPA) onvoldoende zijn voor bescherming van de openbare gezondheid en dat het bureau onverwacht strengere maatstaven tegen vervuiling moet aanleggen.

Dr. Robert Harris, mededirecteur van het Fonds, zei op een persconferentie dat 'Amerika lijdt aan een kankerepidemie. Desondanks hebben onze instellingen die belast zijn met de bescherming van de openbare gezondheid in feite niets gedaan om de men:en tegen kankerverwekkende stoffen te behoeden'.

Hierbij doet het onderzoek, dat voor de EPA is verricht, een aan de door het Amerikaanse bureau voor bescherming van het milieu voorgestelde strengere maatstaven voor melioratie van het water vervuild is, alsmede nader onderzoek naar het verband tussen deze vervuiling en ziekten en een formule voor vermindering van het aantal fluoriden in het water.

Het fonds zegt dat al deze voorstellen onontoeikend zijn en wil dat het Hof van Beroep de EPA bevel geeft ze te herzien. De groep onderhoudt nauw contact met de 'kampioen van de consumentenbelangen' Ralph Nader en beweert 50.000 leden te hebben. Vorig jaar trok het de aandacht met een rapport dat wees op een mogelijk verband tussen het gehalte aan lood in het drinkwater van Philadelphia, Cincinnati, Miami, New Orleans en Miami evenveel verontreinigingen in Delaware, Cincinnati uit de Ohio en Miami uit kleine ondergrondse meren.

Het fonds zegt dat de EPA niets bepaald heeft ten aanzien van carcinogenen, te weinig watermonsters laat nemen om het gehalte aan lood en cadmium vast te stellen en geen maxima heeft gesteld voor het gehalte aan giftige bestanddelen, zoals asbest.

(Retter).