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

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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

— the 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 causative 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 $0{,}250\ m^3/$person/d. and for the important pollutants BOD, N and P, respectively, $w_B = 50$ to 70 g BOD/person/d.; $w_N = 10$ to 15 g N/person/d.; $w_P = 3$ to 6 g P/person/d.
The total flow Q (m^3/d) and waste loads W_i (g/d) are then estimated from the number of population equivalents p by $Q = p \cdot q$ and $W_i = p \cdot w_i$, whereas the pollutant concentrations c_i (g/m^3) are given by $c_i = w_i/q$.

2.2. *Treatment Response*

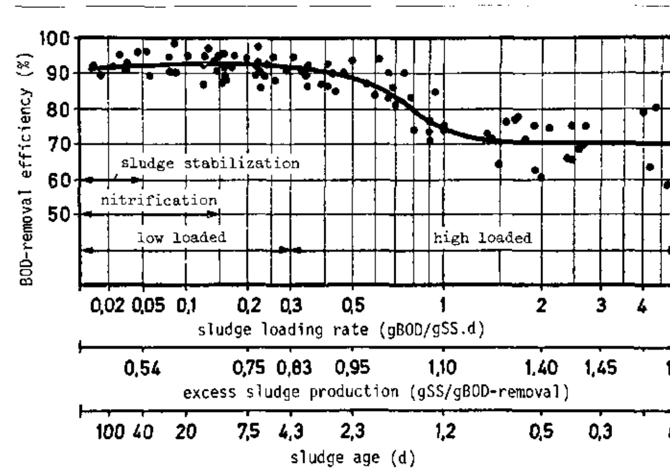
2.2.1. Primary Sedimentation

Most of the settleable solids are removed by primary sedimentation which is designed on the basis of the hydraulic retention time τ in the settling tank (e.g. 2 to 4 hours), and the hydraulic surface loading rate (m^3 sewage per m^2 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 (activited sludge and trickling fiter) 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: $M = V \cdot G$, or by the volatile fraction thereof: $M' = V \cdot G_V$. The F/M-ratio then becomes $F/M = F/V \cdot G$ (g BOD/g solids . d) or $F/M' = F/V \cdot G_V$ (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/m^3 filter volume . d).
Figures 1 and 2 show typical correlations between the F/M ratio (as defined above) and the BOD removal effeciency η (%) for activated sludge and trickling filter

Fig. 1 - The influence of the sludge loading rate on the BOD-removal efficiency [10].



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/m^3 . 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 R_s (kg solids per m^2 tank surface area per hour) and/or the solids volume surface-loading rate R_v (m^3 of sludge solids per m^2 tank surface area per hour). Both parameters are linked by the sludge volume index SVI (m^3 sludge volume per kg solids after 30 min, settling):

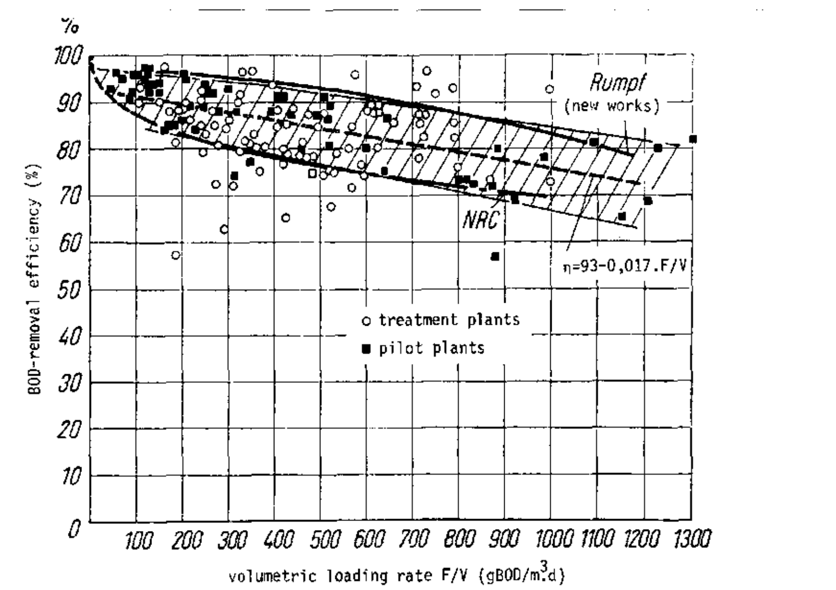


Fig. 2 - The influence of the volumetric loading rate on the BOD-removal efficiency [11].

TABLE 1 - *Organic loading rates (F/M) and expected BOD removal efficiencies.*

System	organic loading rate: activated sludge g BOD/g solids . day	organic loading rate: trickling filter g BOD/m^3 . day	efficiency %
high load	> 1,0	> 800	70 to 80
medium loaded	0,5 to 1,0	400 to 800	80 to 90
low loaded	0,1 to 0,5	200 to 400	85 to 95
sludge stabilization	< 0,1	< 200	90 to 95

$R_v = R_s$. SVI. Typical values for R_s are 2,0 to 3,5 kg solids/m^2 . h for an index of SVI = 0,1 m^3/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 wastewater 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 bij 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 incorporation into the excess sludge is expected to decrease with decreasing loading rates. At fairly low loading rates nitrification of ammonia and, finally, nitrogen removal 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

$$M = (\Sigma X)/n \qquad (1)$$

and of the standard deviation

$$s = \sqrt{\frac{\Sigma (X - M)^2}{n - 1}} \qquad (2)$$

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

$$CV = s/M \qquad (3)$$

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).*

Plant	Type	Flow	Dry solids in aeration tank	Sludge loading rate	Volumetric loading rate	Retention time in aeration tank	Retention time in secondary clarifier	Samples type	Samples number
		10^3 m³/d	kg/m³	kg BOD / kg solids . d	g BOD / m³ . day	h	h	—	—
Cleveland Easterly, Ohio, USA	AS	370	1,25	0,24	300	7,5	3,0	monthly	108
Cranston, R.I., USA	AS	16,4	1,40	0,53	740	5,7	2,0	monthly	108
Gary, Ind., USA	AS	112	3,30	0,12	400	5,0	2,0	monthly	84
Heilbronn, Germany	AS	6,9	2,20	0,55	1200	3,2	4,2	daily	72
Maple Lodge, England	AS	93	4,60	0,10	460	12,0	4,5	daily/monthly	245/193
Marion, Ind., USA	AS	22,5	2,60	0,20	520	4,0	2,0	monthly	71
Slough, England									
first stage	AS	17,3	2,40	0,60	1440	7,2	3,5	daily/monthly	307/121
second stage	TF	17,3			135				

AS = conventional activated sludge plant
TF = trickling filter plant

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

Plant	Parameter	Dimension	Mean of plant input	Mean of plant output	Variation of plant input	Variation of plant output	Correct freq. distribution input	Correct freq. distribution output
Heilbronn	BOD	g/m³	160,6	15,8	43,5	59,6	x	—
			143,1	13,4	1,683	1,774	—	x
Maple	BOD	g/m³	219,9	16,8	20,4	39,1	—	—
Lodge			215,4	15,4	1,225	1,468	x	x
		kg/d		1576		39,9		—
				1466		1,463		x
	flow	m³/d	94262		10,7			—
			93735		1,112			—
Slough	BOD	g/m³	437,7 80,5	27,0	24,2 33,0	33,2	— —	—
			423,7 75,9	25,4	1,304 1,423	1,419	— x(a>0)	x(a>0)
		kg/d		480		46,6		
				428		1,637		x(a>0)
	susp.	g/m³		18,4		43,2		—
	solids			16,9		1,494		x
		kg/d		327		55,4		—
				285		1,697		x
	NH_4^+-N	g/m³		9,4		62,9		—
				7,5		2,057		x(a>0)
		kg/d		175		73,2		—
				126		2,431		x(a>0)
	NO_2^--N	g/m³		0,82		38,8		—
				0,76		1,485		—
		kg/d		13,7		37,1		—
				12,8		1,467		x(a>0)
	NO_3^--N	g/m³		14,5		34,7		—
				13,6		1,447		—
		kg/d		241		31,3		—
				229		1,395		—
	flow	m³/d	17279		22,6			—
			16829		1,262			x(a>0)

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

$$M_l = \exp\left[(\Sigma \ln X)/n \right] \quad (4)$$

and the log standard deviation

$$s_l = \exp\left(\frac{\Sigma(\ln X - \ln M_l)^2}{n-1}\right)^{1/2} \quad (5)$$

Obviously, lnX is then normally distributed. This transformation can cause a negative skewness. A third type of 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 op samples should be taken. The possibilities are

— grab samples, which include all the variation;

— composite samples, taken over a certain avering 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. man
2. standard deviation
3. skewness (g_1)
4. curtosis (g_2)

from

a. the original data

b. the log-transforms of the data.

The results are summarized in table 3 (analysis of 'daily data') and table 4 (analysis of 'montly 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 municipal wastewater treatment plant parameters in supplemented by results from treating

— municipal wastewater at Houston, Texas (table 5);

— petrochemical wastewater (table 6);

— paper mill wastewater (table 7).

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_l) 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 con-cusion 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 4 - *Frequency Analysis of 'monthly data (1).*

Plant	Parameter	Dimension	Mean of plant input	Mean of plant output	Vatiation of plant input	Vatiation of plant output	Correct freq. distribution input	Correct freq. distribution output
Cleveland	BOD	g/m³	95,6	11,5	25,2	36,0	—	x
			92,6	10,6	1,290	1,529	x	—
Cranston	BOD	g/m³	176,7	26,9	32,2	59,1	—	—
			168,1	23,3	1,370	1,700	x	x
Gary	BOD	g/m³	84,4	12,8	25,6	64,2	—	—
			81,9	10,7	1,277	1,806	x	x
Maple	BOD	g/m³	370,2	16,2	16,4	42,2	—	—
Lodge			365,3	14,8	1,176	1,536	x	x
	BOD	kg/d	31890	1330	36,2	46,8	—	—
			29720	1206	1,472	1,554	(x)	x
	flow	m³/d	85750		31,0			x
			81350		1,400			—
Marion	BOD	g/m³	85,1	8,4	33,7	49,4	—	—
			81,1	7,5	1,359	1,626	x	x
Slough	BOD	g/m³	508,6	23,5	14,4	20,8	—	—
			503,4	23,0	1,154	1,232	x	x
	BOD	kg/d	15840	742	15,7	27,9	x	—
			15640	715	1,178	1,308	—	x
	flow	m³/d	31430		15,7			—
			31060		1,168			x

Explanation:
There are 2 rows of results for each parameter:
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— 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.

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

Plant and data	n	coefficient of variation CV (%)	log-normal standard deviation s_l
Southwest			
daily	287	84	2,07
monthly	23	45	1,54
Northeast			
daily	100	152	2,99
monthly	23	45	1,54
Sagemont			
daily	51	79	2,00
monthly	12	43	1,50

All daily date are log-normally distributed.
s_l is estimated from CV.

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

Plant and samples	influent	effluent
Complete mix activated sludge, F/M $\simeq$ 0,1; $\tau \simeq$ 1 day weekly composite soluble COD samples	CV = 38 % s_l = 1,44	CV = 84 % s_l = 2,08
Complete mix activated sludge, F/M $\simeq$ 0,2; $\tau \simeq$ 2,5 days daily compositie BOD samples	CV = 33 % s_l = 1,38	CV = 65 % s_l = 1,81
Facultative ponds in series $\tau \simeq$ 20 days, 3 day composite BOD samples		CV = 31 % s_l = 1,35

s_l is estimated from CV

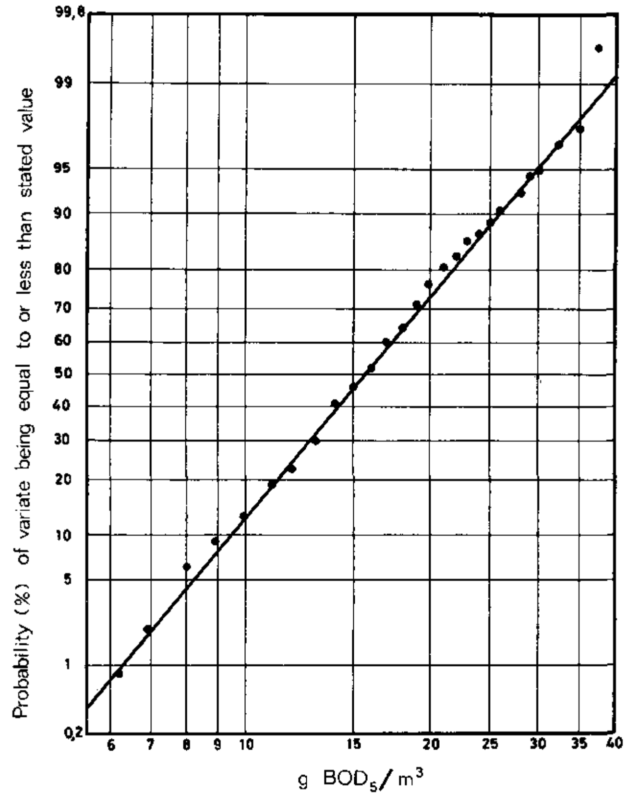


Fig. 3 - Log-probability of the daily effluent BOD data of Maple Lodge [1].

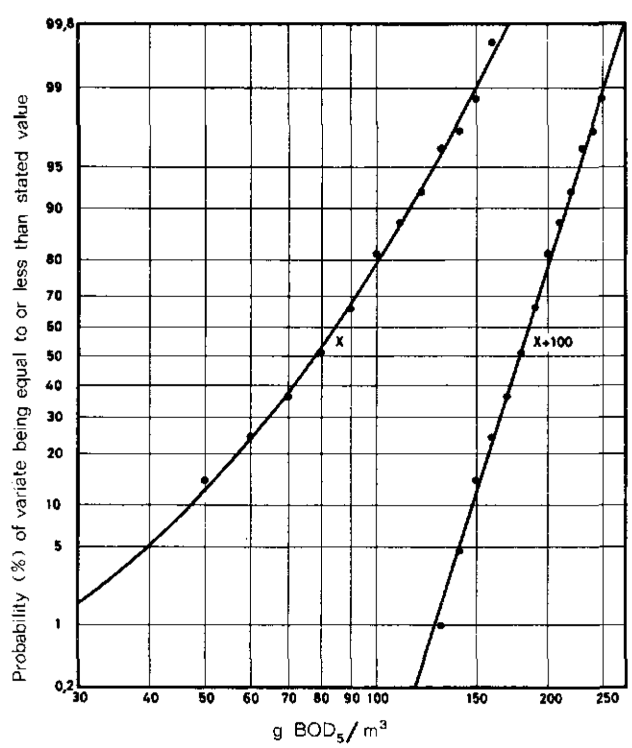


Fig. 4 - Log-probability plot of the first stage effluent BOD data of Slough and skewness correction by addition of a = 100 [1].

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 tot 1,468, whereas the standard deviation is reduced from s = CV . M = 0,204 . 219,9 = 44,9 to 0,391 . 16,8 = 6,57 g BOD/m^3, a reduction of 85 %. Thus the absolute magnitude of the variations is reduced by treatment, the relative variability (CV, s_l) 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

$$s_{l,m} = s_{l,d}{}^{1/\sqrt{n_m}} \quad (7)$$

From s_l = 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 $s_{l,m} = 1{,}468^{1/\sqrt{20,4}} = 1{,}089$, whereas the actual value (table 4) is 1,536.
The significant differences between 'expected' and actual values of the CV and s_l 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 efficiences;

— season dependent wastewater temperatures effecting season dependent treatment efficiencies;

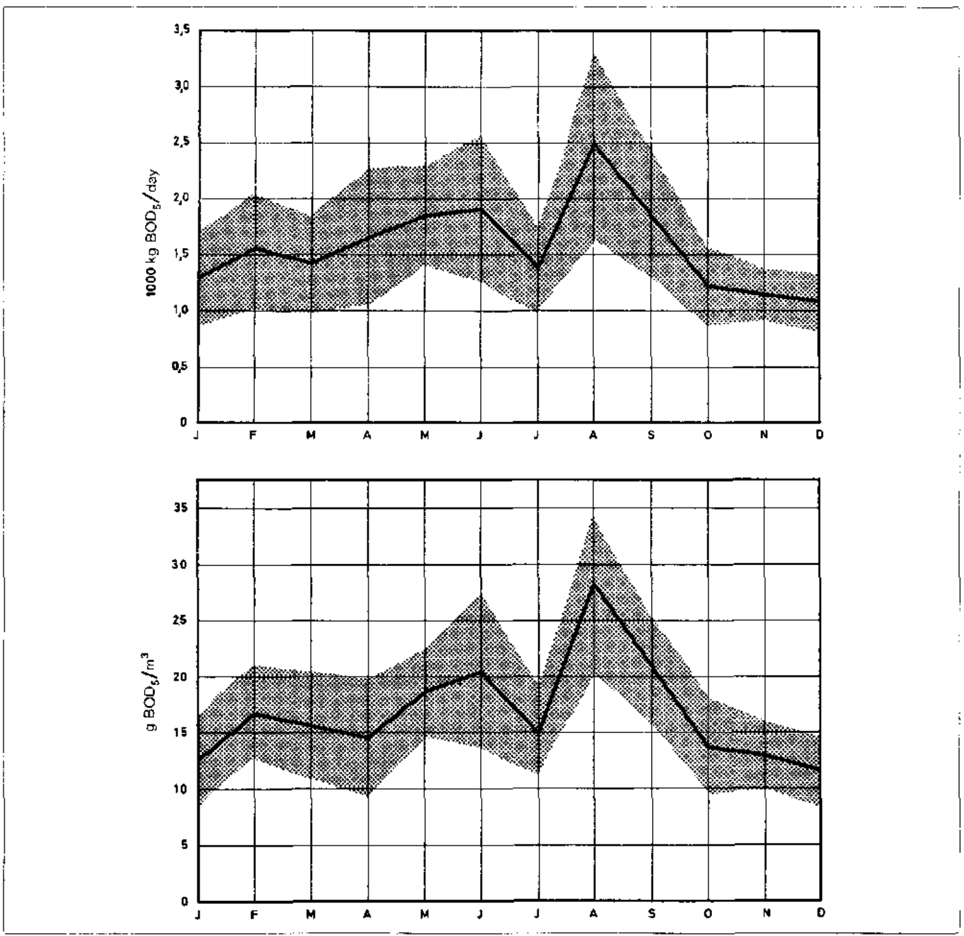
— season dependent waste production, brought about by the pattern of life of the

TABLE 7 - *Variability of effluent loads (kg/d) from 11 paper mill wastewater treatment plants, based on daily composite samples (after [3]).*

nr.	Product	Treatment	τ(d)	Parameter	n	CV (%)	s_l	type of distribution
1.	Groundwood	AST		BOD	179	67	1,84	log-normal
				SS	223	164	3,13	log-normal
2.	Sulfite	ASB	11	BOD	325	35	1,40	log-normal
				SS	327	64	1,80	normal
3.	Sulfite	ASB	9	BOD	333	26	1,29	—
				SS	333	35	1,40	log-normal
4.	Bleached kraft	ASB + PS	17 + 7	BOD	323	66	1,82	log-normal
				SS	267	68	1,85	log-normal
5.	Bleached kraft	ASB + PS	2 + 12	BOD	299	79	2,01	log-normal
				SS	187	94	2,22	log-normal
6.	Bleached kraft	ABS + PS	14 + 60	BOD	289	80	2,02	log-normal
				SS	290	58	1,71	—
7.	Bleached kraft	AST + PS		BOD	124	55	1,67	log-normal
				SS	273	42	1,50	normal
8.	Bleached kraft	ASB	23	BOD	216	37	1,43	normal
				SS	214	43	1,51	—
9.	Bleached kraft	AST		BOD	326	46	1,55	log-normal
				SS	329	68	1,85	log-normal
10.	Bleached kraft	AST		BOD	328	63	1,78	log-normal
				SS	328	74	1,94	log-normal
11.	Deinked	ASB	7	BOD	173	53	1,64	normal
				SS	326	97	2,26	log-normal

SS suspended solids
AST activated sludge treatment
ASB aerated stabilization basin treatment
PS post storage
s_l is estimated from CV

Fig. 5 - Monthly averages and standard deviations from daily samples of the Maple Lodge effluent BOD [1].



population and of industrial production.

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/m^3 (0,367);
3. yearly change of daily sewage flow: + 5300 m^3/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/m^3 (— 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 η is obtained for plug flow:

$$\eta_{PF} = 1 - e^{-k\tau} \tag{8}$$

complete mixing flow:

$$\eta_{CM} = \frac{k\tau}{1 + k\tau} \tag{9}$$

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):

$$F = F_c + F_p \cdot \sin(\omega t) \tag{10}$$

where

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

Typical input periods for municipal wastewater are

— τ_p = 1 day to describe the hourly input variation
— τ_p = 7 days to generalize possible week-day variations
— τ_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:

$$c = \frac{F_c}{Q} \cdot e^{-k\tau} + A_{PF} \cdot \frac{F_p}{Q} \cdot \sin(\omega t - \Theta_{PF}) \tag{11a}$$

with

A_{PF} reduction factor of input amplitude

$$A_{PF} = e^{-k\tau} = 1 - \eta_{PF} \tag{11b}$$

Θ_{PF} phase shift

$$\Theta_{PF} = \omega \cdot \tau \tag{11c}$$

b. complete mixing flow

$$c = \frac{F_c}{Q} \cdot \frac{1}{1 + k\tau} + A_{CM} \cdot \frac{F_p}{Q} \cdot \sin(\omega t - \Theta_{CM}) \tag{12a}$$

with

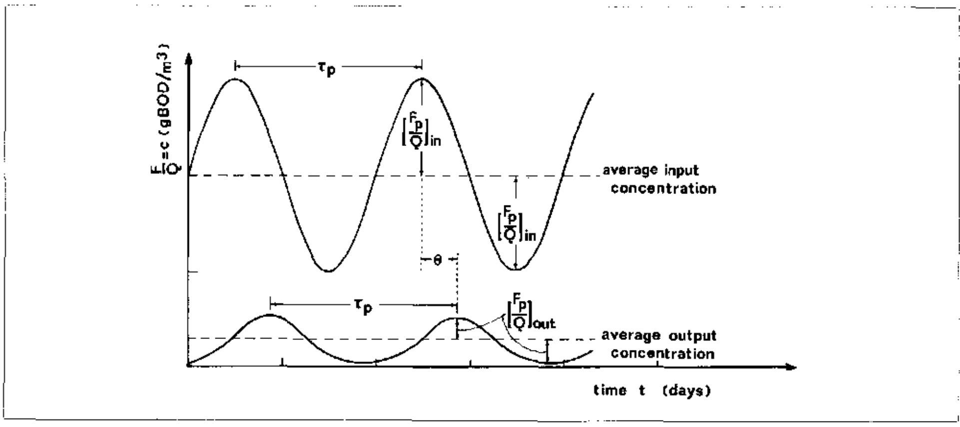
$$A_{CM} = [(1 + k\tau)^2 + (\omega\tau)^2]^{-1/2} = [\frac{1}{(1 - \eta_{CM})^2} + (\omega\tau)^2]^{-1/2} \tag{12b}$$

$$\Theta_{CM} = \text{arctg} \frac{\omega\tau}{1 + k\tau} = \text{arctg} [\omega\tau \cdot (1 - \eta_{CM})] \tag{12c}$$

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 η_{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 Θ_{PF} in terms of time is equal to the retention time τ.

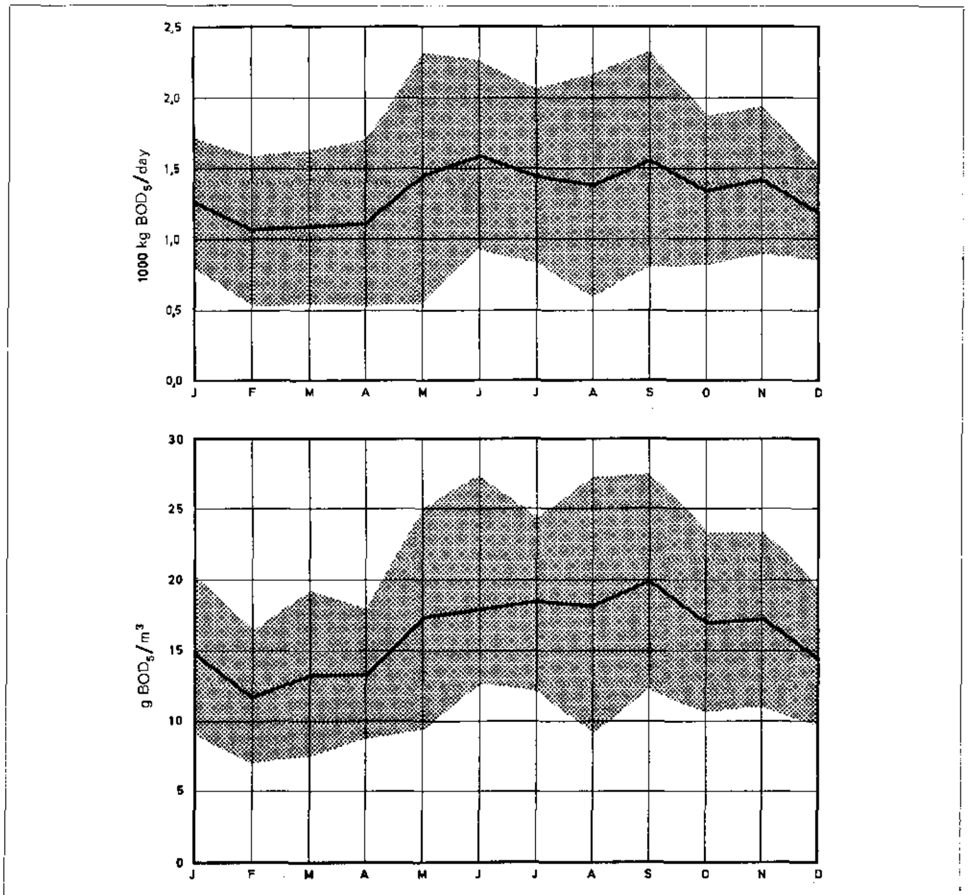
The response of a complete mixing system (rectangular or round activated sludge tank, oxidation ditches) to a periodic input is different; the amplitude is reduced by degradation (as before) and dilution or equalization. The first and second terms of equation 12b quantify these mechanisms, respectively.

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.

Fig. 6 - Monthly averages and standard deviations from monthly samples of the Maple Lodge effluent BOD [1].



such basins, k and/or η_{CM} are zero, hence

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

$$\Theta'_{CM} = \mathrm{arctg}\,(\omega\tau) \tag{13b}$$

To illustrate the effect of complete mixed activated-sludge systems and equalization basins on the amplitude of a periodic concentration input, the factor A_{CM} is computed as a function of the input frequency ω for three hypothetical systems:

a. retention time τ = 0,25 days, η = 85 %, a medium loaded plant;

b. retention time τ = 0,5 days, η = 90 %, a low loaded system;

c. retention time τ = 2 days, η = 95 %, an extended aeration system with sludge stabilization.

The results are summarized in figure 7 by plotting log A_{CM} versus log $(1/\tau_p)$ 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_{CM}, the difference between $A_{CM} = 1$ 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_{CM} versus log (τ/τ_p), i.e. versus the logarithm of the ratio of retention time τ over periodic time τ_p. Again equalization and decay effects can be differentiated.
It is clear from figures 7 and 8 that equalization effects prevail at high frequency input (short τ_p or large τ/τ_p). This is true when $\tau/\tau_p > 2$, or when $\tau < 2 \,.\, \tau_p$. Dilution effects diminish when the input frequency decreases and decay effects become more important. When $\tau/\tau_p < 0{,}05$ or when $\tau_p > 20 \,.\, \tau$, 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 τ is significantly longer than 0,05 times the input period time τ_p.
The periodic input, modelled before, may be superimposed by other input functions like:

— a pulse function (e.g. a shock load of short duration);

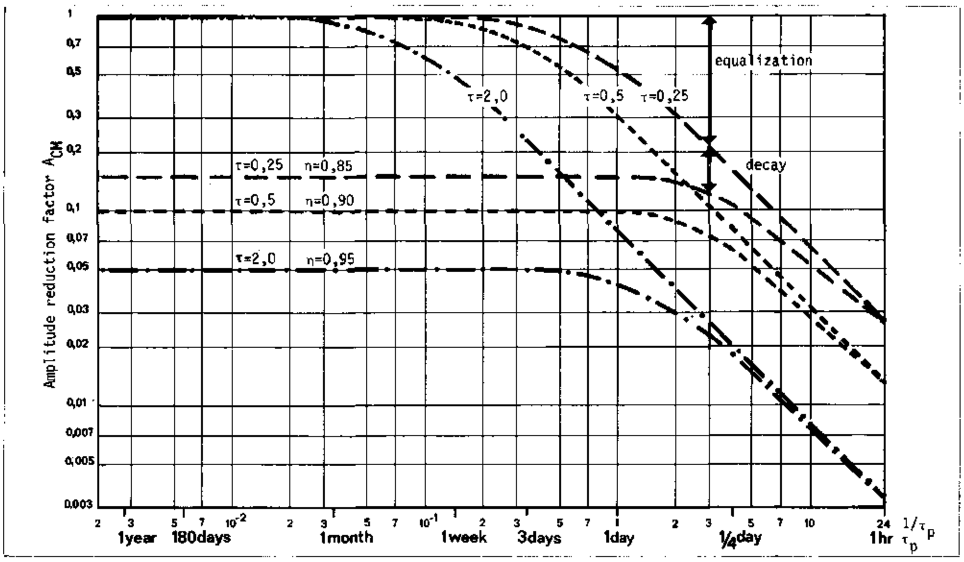


Fig. 7 - The influence of retention time τ and treatment efficiency η on the amplitude reduction factor A_{CM} at various input periods τ_p.

— 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. 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/s_i of a complete mixing system is given by

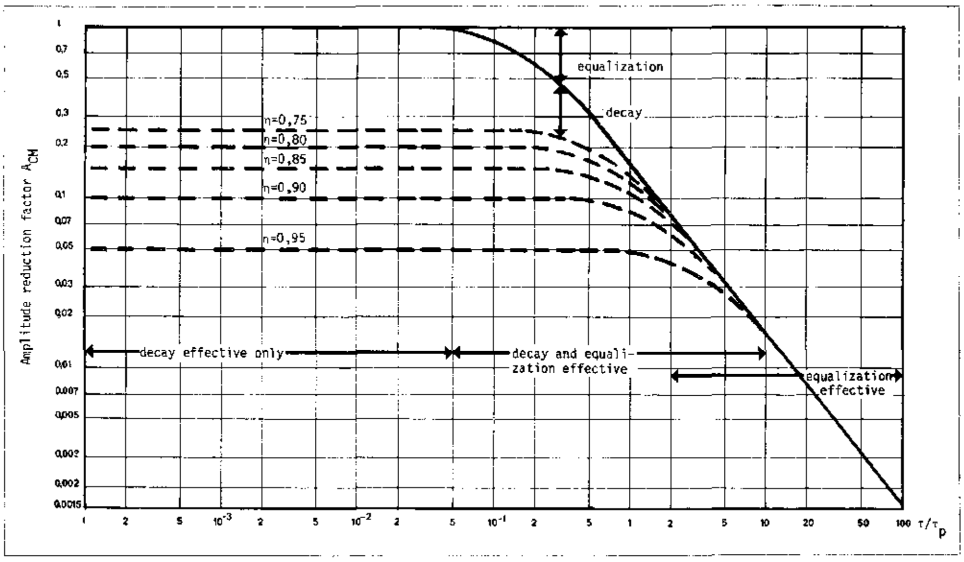
$$\frac{s_e}{s_i} = \sqrt{\frac{\tau_s}{2\,.\,\tau\,.\,(1 + k\tau)}} \tag{14}$$

$$= \sqrt{\frac{\tau_s\,.\,(1 - \eta)}{2\,.\,\tau}}$$

where τ_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 (τ). 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

Fig. 8 - The effect of treatment efficiency η and τ/τ_p-ratio on the amplitude reduction factor A_{CM}.



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

$$\frac{CV_e}{CV_i} = \frac{s_e}{s_i} \cdot \frac{M_i}{M_e} = \frac{1}{1-\eta} \cdot \sqrt{\frac{\tau_s \cdot (1-\eta)}{2 \cdot \tau}} \qquad (15)$$

$$= \sqrt{\frac{\tau_s}{2 \cdot \tau\,(1-\eta)}}$$

The above ratio is always greater than one, except for $\tau_s < 2 \cdot \tau \cdot (1-\eta)$. At very short averaging periods of sampling (τ_s), 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 micro-organisms 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 1/d; G_V = 2200 g/m³, τ = 0,88 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

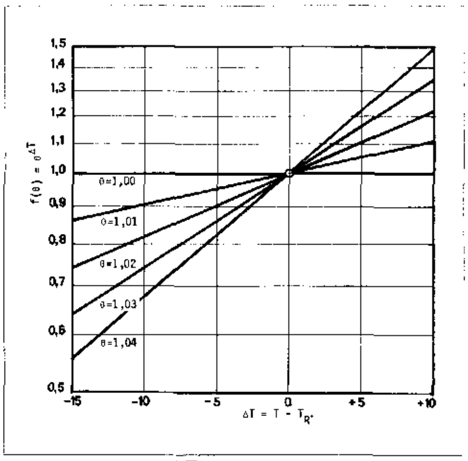
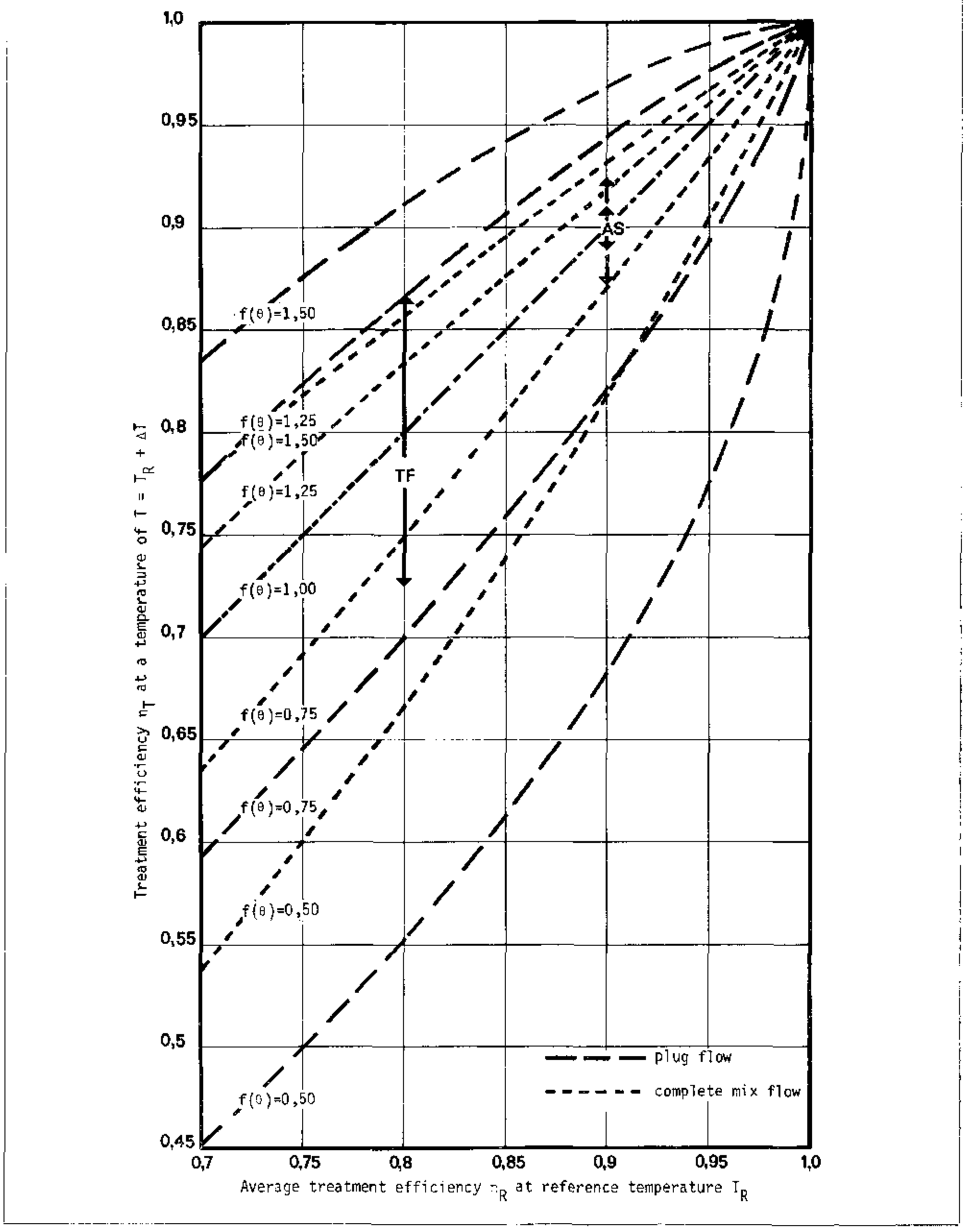


Fig. 9 - $f(\theta) = \theta^{\Delta T}$ as a function of ΔT.

TABLE 8 - *Effect of influent concentration variability on the treatment efficiency and effluent variability (data after [4]).*

Parameter	constant feed	variable feed
Influent BOD (g/m³)		
average	1400	1400
standard deviation	120	850
coefficient of variation	8,6 %	60,7 %
Effluent BOD (g/m³)		
average	22	180
standard deviation	16	240
coefficient of variation	73 %	133 %
BOD removal efficiency	98,4 %	87,1 %
k (acc. eq. 9)	70,9 1/d	7,67 1/d
s_e/s_i - ratio	0,133	0,282
predicted (eq. 14, τ_s = 1 d)	0,106	0,271
CV_e/CV_i - ratio	8,48	2,20
predicted (eq. 15, τ_s = 1 d)	6,00	2,10

Fig. 10 - The effect of average treatment efficiency η_R (at T_R), temperature coefficient θ, and temperature difference ΔT, on the treatment efficiency η_T (at T).



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

$$k_T = k_R \cdot \Theta^{T - T_R} = k_R \cdot \Theta^{\Delta T} = k_R \cdot f(\Theta) \qquad (16)$$

where

k_T = reaction coefficient at temperature T

k_R = reaction coefficient at reference temperature T_R

Θ = 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 η_T at temperature T and η_R at the reference temperature T_R for

plug flow systems

$$\eta_T = 1 - (1 - \eta_R)^{f(\Theta)} \qquad (17)$$

complete mixing systems

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

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 ΔT. Within the normal wastewater temperature variation and the cited range of Θ, $f(\Theta)$ 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 η_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

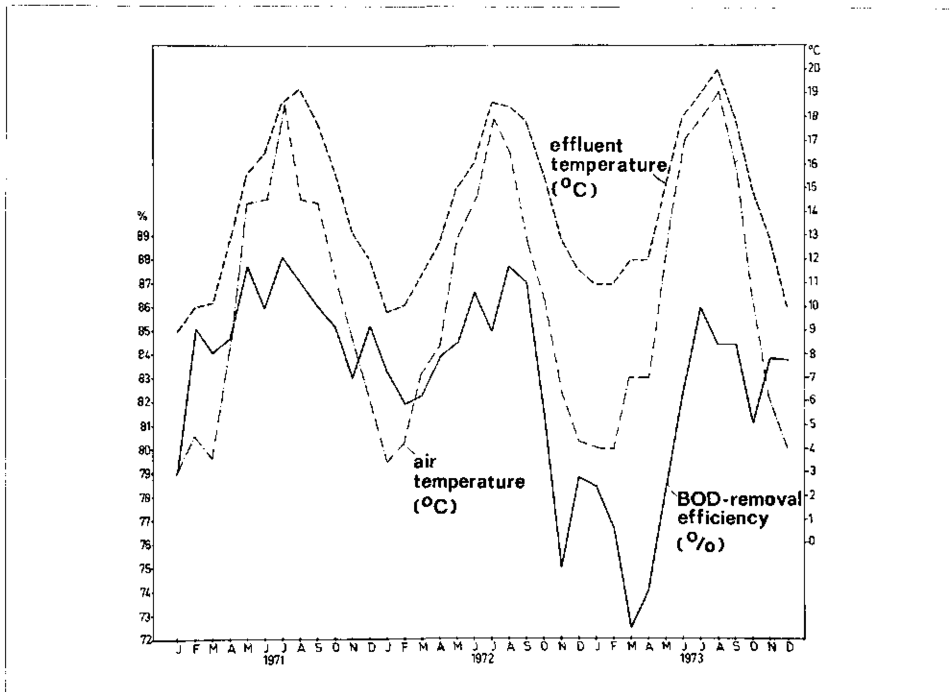


Fig. 11 - The influence of wastewater temperature on the BOD-removal efficiency at the Utrecht trickling filter plant [8].

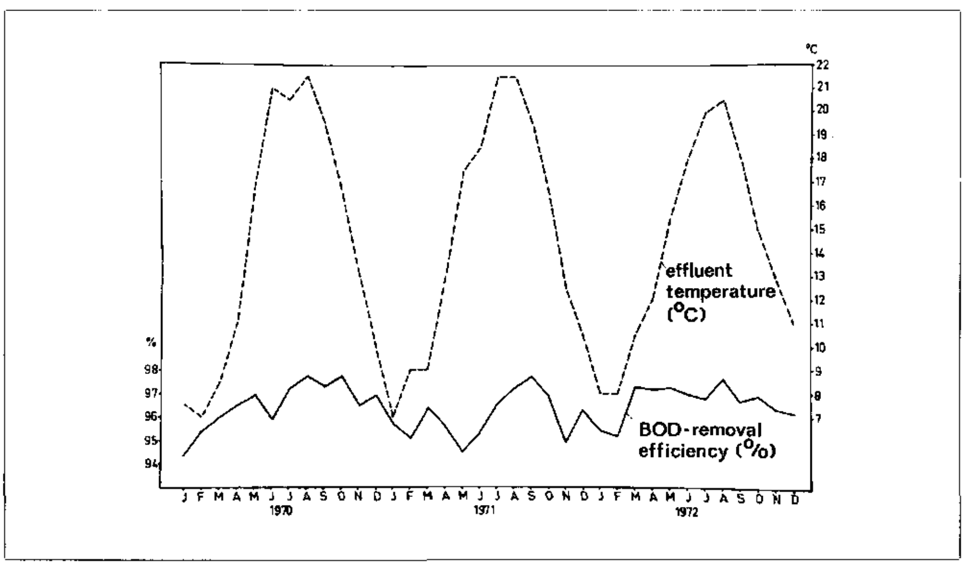


Fig. 12 - The influence of wastewater temperature on the BOD-removal efficiency at the Amsterdam activated sludge plant [8].

— 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_R) of $\Delta T = -7{,}5$ °C (winter) and $+7{,}5$ °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 Θ and the temperature change ΔT, but also on

— the average treatment efficiency η_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 = ± 7,5 °C, total temperature range hence 15 °C) the average treatment efficiency η_R of a system has a much greater influence on its temperature sensitivity than its temperature coefficient Θ. This is seen from a comparison of the large η_T-range of the trickling filter ($\eta_R = 0{,}8$; $\Theta = 1{,}030$) and the relatively small η_T-range of the activated-sludge system ($\eta = 0{,}9$; $\Theta = 1{,}015$) in figure 10.

The last mentioned effect is clearly illustrated by data [8] from the trickling-filterplant at Utrecht (fig. 11) and from the activated-sludge plant at Amsterdam (fig. 12). When comparing extreme values of η_T 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 Θ 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 Θ.

3.3. *Conclusions: How to Reduce Effluent Quality Variability*

In the foregoing sections the variability of wastewater treatment system parameters has been decribed 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 effect of changes 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 ($\tau_p < 0{,}1 \cdot \tau$), 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,

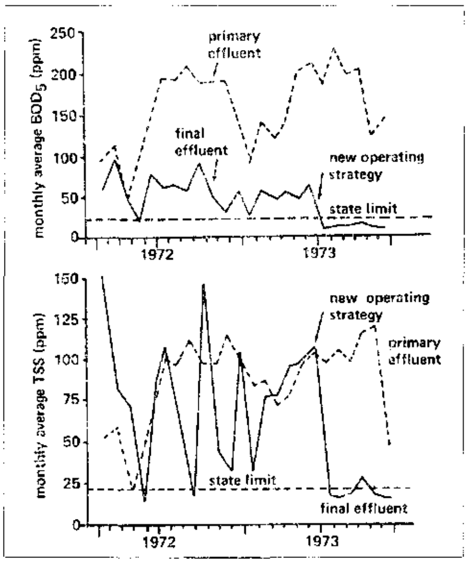


Fig. 13 - The influence of a new operating strategy on the effluent quality at the Hillsboro contact stabilization plant [9].

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 forementioned 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 (5 minutes). By adjusting the return and excess sludge flow, 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.

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Notation

A	—	amplitude reduction factor: effluent/influent amplitude of a periodic function
CM	subscript	parameter based on complete mixing flow conditions
CV	—, %	coefficient of variation
F	g/d	load on a biological treatment system (BOD, N, P)
$f(\Theta)$		$= \Theta^{T - T_R}$
G	g/m^3	activated-sludge suspended-solids concentration
G_V	g/m^3	volatiles of G
k	1/d	first-order reaction coefficient
k_T	1/d	k at temperature T
k_R	1/d	k at reference temperature T_R
l	subscript	parameter based on log-normal distribution
M		biological mass active in degrading organics
M	variable	mean
n	—	number of samples
p	—	number of population equivalents
PF	subscript	parameter based on plug flow conditions
Q	m^3/d	wastewater flow rate
q	m^3/person d	specific wastewater flow rate: m^3 per person daily
R_s	$kg/m^2 . h$	solids surface-loading rate
R_v	$m^3/m^2 . h$	solids volume surface-loading rate
s	variable	standard deviation
SVI	m^3/kg	sludge volume index
t	d	time
T	°C	temperature
T_R	°C	reference temperature
W	g/d	raw waste load (BOD, N, P)
w	g/person d	specific raw waste load in g per person daily
V	m^3	reactor tank volume
X	variable	quality parameter
$\triangle T$	°C	temperature difference: $T - T_R$
η	—, %	(BOD) removal efficiency
η_R	—, %	η at reference (average) temperature
η_T	—, %	η at temperature T
Θ	—	temperature coefficient
Θ	—	phase shift
τ	d	reactor retention time $= V/Q$
τ_p	d	time period of a periodic function
τ_s	d	averaging time of sampling
ω	1/d	frequency of a periodic function $\omega = 2\pi/\tau_p$