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DENITRIFICATION BY PACKED BED REACTORS IN THE PRESENCE OF CHROMIUM(VI)

RESISTANCE TO INHIBITION

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Abstract—The effect of chromium Cr^{6+} on bacterial denitrification was investigated. The long-term influence of chromium presence was observed in packed bed reactors using methanol, ethanol, *n*-propanol, *n*-butanol, sec-butanol, tert-butanol, iso-butanol and *n*-pentanol as the carbon and energy sources for denitrifiers. Short-term influence was investigated by the inhibition coefficient K_i determination within activated sludge under anoxic conditions. The measured inhibition constant K_i was equal to 84.2 mg l^{-1} Cr^{6+}, independently of the kind of organic compound utilized as the electron donor for the bacterial system. The concepts of the reactor resistance to inhibition (RRI) and the resistance to inhibition (RI) have been evaluated.

Key words—denitrification, inhibition, chromium Cr^{3+}

INTRODUCTION

Chromium in water can exist in a number of forms from Cr^{3-} to Cr^{6+} (Smillie *et al.*, 1981) but according to Schroeder and Lee (1975) only Cr^{3+} and Cr^{6+} are environmentally important. Chromium salts are soluble and toxic for aquatic organisms, especially salts of chromium Cr^{6+} (Mearns *et al.*, 1976). The influence of toxic compounds on the denitrification process is not very well recognized (Christensen and Harremoes, 1977). In this work an attempt to estimate the influence of chromium Cr^{6+} on the denitrification process in relation to various compounds used as the energy and carbon sources was made. The matter of interest was the difference between the short- and long-term influence of chromium on the process course. During the long-term process some adaptation mechanisms of the bacteria involved may be expected and results can be different from those obtained in unacclimated systems under shock-load conditions. For the long term influence investigations the packed bed reactor (PBR) was chosen because of the long residence time of the microorganisms within the reactor. The short term influence was investigated in activated sludge under anoxic conditions. The effect of chromium in the case of the PBR was observed as the relation between the chromium concentration and the nitrate removal rate. In the case of the denitrification in activated sludge the inhibition coefficient K_i was determined. Utilization of various compounds as the electron donors allowed observation of the effect of chromium concentration on the denitrification efficiency for each kind of substrate oxidized.

MATERIALS AND METHODS

The packed bed reactors used for the long-term influence of chromium investigations consisted of plastic columns 45 mm i.d. and 0.8 m tall. Columns were placed in a thermostatic bath and a temperature of 25°C was maintained during the whole experimental period. Reactor packing consisted of ceramic Raschig rings 15 mm long, 16 mm o.d. and 10 mm i.d. The PBR were filled with the packing material up to 0.5 m height. Total surface of the packing was 0.27 m^2. The porosity of the reactors was 0.625, total volume 0.8 l. and liquid volume 0.5 l. Detention time, based on the liquid volume was 1 h. The wastes to be treated were supplied to the columns with the use of peristaltic pumps and entered at the bottom. The effluent ports were placed at the top of the reactors. The packing material was submerged during the experiments. The synthetic waste was prepared using tap water and stored in plastic containers, feed solutions were changed twice a week. The nitrate concentration in the influent was adjusted to 20 mg l^{-1} NO_3-N (as KNO_3), phosphate to 2 mg l^{-1} PO_4-P (as Na_2HPO_4). The organic compounds: methanol, ethanol, *n*-propanol, *n*-butanol, sec-butanol, tert-butanol, iso-butanol and *n*-pentanol were added to the individual containers in the concentrations of about 100% greater than stoichiometric requirements. The chromium concentration Cr^{6+} (added as $K_2Cr_2O_7$) was raised during the experiment from 1 mg l^{-1} Cr^{6+} at the very beginning of the process until progressive deterioration of the results in terms of denitrification efficiency was obtained. The feed solutions were deoxygenated by means of sodium sulphite Na_2SO_3 by stoichiometric addition according to the dissolved oxygen concentration. All the reagents used were laboratory grade. The stoichiometric quantities of the compounds used as the electron donors were obtained from the following equations:

for methanol:

$$NO_3^- + \tfrac{5}{6}CH_3OH = \tfrac{1}{2}N_2 + \tfrac{5}{6}CO_2 + \tfrac{7}{6}H_2O + OH^- \quad (1)$$

for ethanol:

$$NO_3^- + \tfrac{5}{12}C_2H_5OH = \tfrac{1}{2}N_2 + \tfrac{5}{6}CO_2 + \tfrac{1}{4}H_2O + OH^- \quad (2)$$

for propanol:

$$NO_3^- + \tfrac{10}{35} C_3H_7OH = \tfrac{1}{2} N_2 + \tfrac{5}{6} CO_2 + \tfrac{31}{16} H_2O + OH^- \quad (3)$$

for butanol:

$$NO_3^- + \tfrac{5}{24} C_4H_9OH = \tfrac{1}{2} N_2 + \tfrac{5}{6} CO_2 + \tfrac{17}{24} H_2O + OH^- \quad (4)$$

for pentanol:

$$NO_3^- + \tfrac{1}{6} C_5H_{11}OH = \tfrac{1}{2} N_2 + \tfrac{5}{6} CO_2 + \tfrac{37}{30} H_2O + OH^-. \quad (5)$$

The reactor seeding was accomplished by means of activated sludge acclimated to the anoxic conditions and to the other parameters of the experimental conditions. The activated sludge taken from the Municipal Treatment Plant was held under anoxic conditions during 24 h. To the sludge 100 mg l^{-1} NO_3-N, 2 mg l^{-1} PO_4-P, the appropriate amount of organic carbon compound and 1 mg l^{-1} chromium Cr^{6+} were added. The procedure was repeated for each of the carbon compounds used. After 24 h considered as the adaptation period the inoculum was poured into the individual filters and the experiment was started. Synthetic wastes based on tap water containing 1 mg l^{-1} Cr^{6+} were pumped through the filters by means of peristaltic pumps. Samples were collected twice a week. After obtaining good results, which means nitrate conversion to nitrogen gas of about 95% the chromium concentration in the influent was increased to the next level and so on until progressive deterioration of denitrification was obtained. Chromium concentration was measured by means of the AAS spectrometer. Measurements for pH were made by means of a pH meter, for ammonium by means of an ion selective electrode. The nitrate and nitrite were measured by following the procedures outlined in *Standard Methods* (APHA, 1975). Organic carbon concentration was analysed with Beckman Organ Carbon Analyzer. All the colorimetric determinations were made by means of spectrophotometer UV-VIS Perkin–Elmer–Hitaschi 2000.

The short-term influence of chromium Cr^{6+} on the denitrification process was assessed by measuring the inhibition constant K_i in activated sludge under anoxic conditions. The inhibition by chromium Cr^{6+} can be considered as that of a non-competitive kind. The relationship between the reaction velocity and inhibitor concentration for this type of inhibition can be described by the following equation (Aiba *et al.*, 1973):

$$V = \frac{V_{max} \cdot s \cdot K_i}{(K_m + s) \cdot (K_i + i)} \quad (6)$$

where:

V = velocity of reaction, mg l^{-1} h^{-1},
V_{max} = maximum velocity of an enzyme catalyzed reaction when saturated with substrate, mg l^{-1} h^{-1},
s = substrate concentration, mg l^{-1},
i = inhibitor concentration, mg l^{-1},
K_m = Michaelis constant, mg l^{-1},
K_i = inhibition constant, mg l^{-1}.

The value of K_i is equal to the concentration of inhibitor which causes a decrease in reaction rate to half of the maximum reaction rate. The velocity of the biochemical reactions within the activated sludge may be described by the Michaelis–Menten equation:

$$V = \frac{-ds}{dt} = \frac{V_{max} \cdot s}{K_m + s} \quad (7)$$

When the concentration of substrate within the sample is much greater than the Michaelis constant $s \gg K_m$ equation (7) may be reduced to the form:

$$V = V_{max}. \quad (8)$$

Keeping the substrate concentration great enough during the measurement the measured reaction velocity is constantly equal to the maximum reaction velocity and the reaction is that of zero order. On the same basis equation (6) can be reduced to the form:

$$V = \frac{V_{max} \cdot K_i}{K_i + i}. \quad (9)$$

A simple graphical method which gives K_i was given by Dixon (Dixon and Webb, 1964). If the reaction velocity is determined with a series of inhibitor concentrations a linear relationship is obtained on plotting $1/V$ against i. The point of intersection with the baseline gives $-K_i$ directly. This is evident from linearization of the equation (9):

$$\frac{1}{V} = \frac{1}{V_{max}} + \frac{i}{V_{max} \cdot K_i}. \quad (10)$$

By putting $1/V = 0$ the intersection point with the baseline gives $-K_i$. Measurements for K_i were made in activated sludge under anoxic conditions. Activated sludge taken from the Municipal Treatment Plant was aerated continuously during 24 h for oxidation of the organic compounds adsorbed on the flocs and dissolved in the sludge liquor. After this the sludge was three times washed, suspended in tap water and transferred into 1 l. plastic containers which were placed in a thermostatic bath. The measurements were carried out at 25°C. Complete mixing was accomplished by means of magnetic stirrers. Before the measurement the samples were moderately aerated and adapted to the temperature of measurement for 2 h. After 2 h the solutions of 10 g l^{-1} KNO_3, 10 g l^{-1} $K_2Cr_2O_7$ and 10 g l^{-1} of alcohol were added to each of the containers. The nitrate concentration at the beginning of the measurement was about 50 mg l^{-1} NO_3-N. The concentrations of chromium within the samples were different, according to experimental programme and have been presented in Figs 3 and 4. The concentration of alcohol within the samples was 200 mg l^{-1} at the beginning of the measurement. Aeration was stopped and the oxygen consumption was observed by means of an oxygen electrode. The measurements were begun 5 min after obtaining anoxic conditions and the samples were collected every 10 min for 2 h. The reaction in the collected samples was stopped by addition of chloroform and the samples were analyzed for the sum of nitrate and nitrite concentration with the use of an automatic analyzer. The concentration of microorganisms was measured as MLSS. Reaction rate was determined by plotting the sum of nitrate and nitrite concentration versus time and was given in relation to MLSS as the specific reaction rate, k. The reaction rates obtained for different chromium concentrations were next taken for the inhibition constant K_i determination according to the previously described Dixon's method.

RESULTS

The results obtained during the operation of the packed bed reactors supplied with various organic compounds serving as the electron donors for denitrifying bacteria have been presented in Figs 1, 2 and in Table 1. Despite the long running time equal to: in the case of the PBR supplied with methanol 95 days, ethanol—140 days, *n*-propanol—145 days, *n*-butanol—120 days, sec-butanol—82 days, tert-butanol—28 days, iso-butanol—155 days and *n*-pentanol—150 days no adaptation of the bacterial system to the presence of chromium was observed. The chromium concentration was generally fixed in the influent at 1 mg l^{-1} Cr^{6+} at the very beginning of the process and was raised to a few mg l^{-1} Cr^{6+} by the end. In the cases of methanol, sec-butanol and

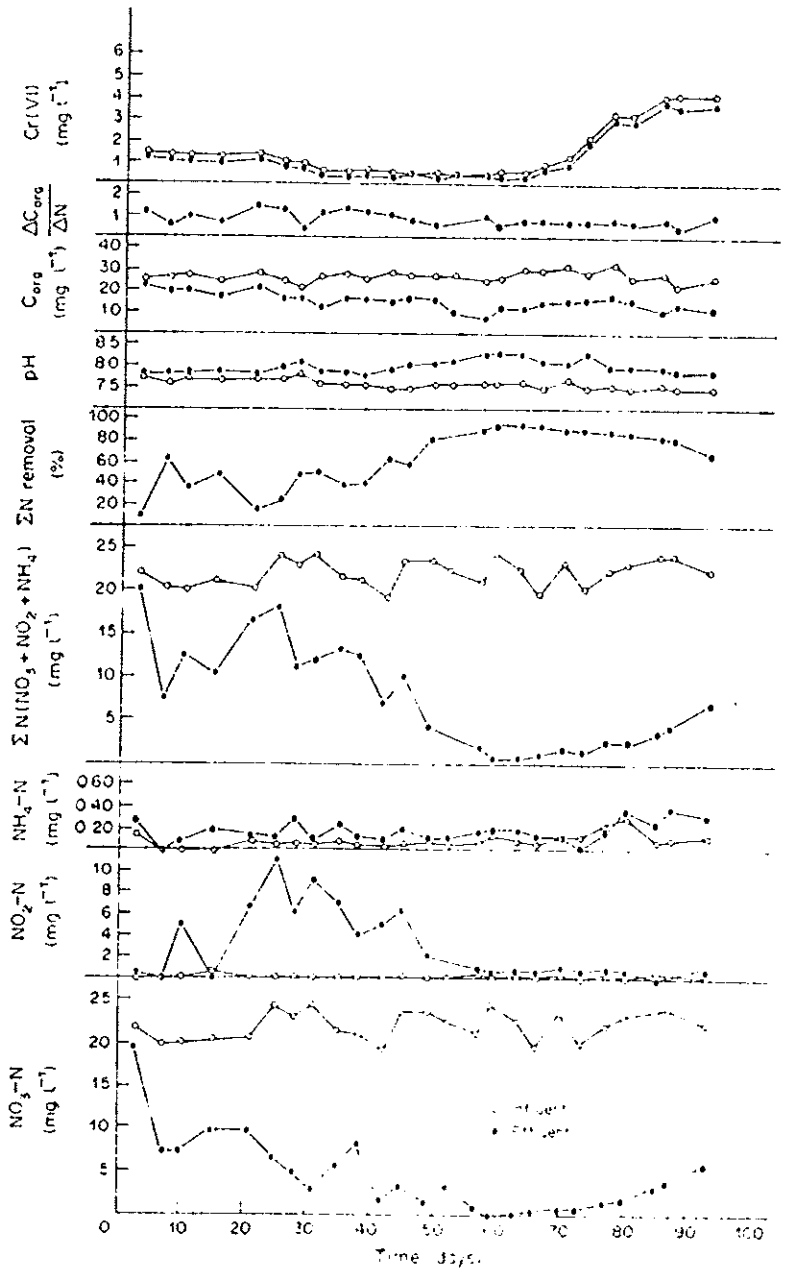

Fig. 1. The performance of the PBR using methanol as the electron donor

tert-butanol the start-up procedures failed with the chromium concentrations in the influent equal to 1 mg l^{-1} Cr^{6+}. Decreasing the chromium concentration to 0.5 mg l^{-1} Cr^{6+} improved the results in the cases of methanol and sec-butanol. In the case of tert-butanol even the removing of chromium from the influent did not allow operation of the reactor with the efficiency close to 100% denitrification. The results in terms of the denitrification efficiency deteriorated with the increase in chromium concentration, independently of the carbon compound utilized. The pH value in the influent was in the range 7.5–7.8 in all cases and increased to 8.5–9.1 in the effluent depending on the process efficiency. In the investigated reactors it was observed that the initial increase in chromium concentration did not result in any decrease in denitrification efficiency. As the criterion of the inhibition starting point the concentration of chromium which caused a decrease in denitrification efficiency below 95% has been accepted. It was assumed that the 5% decrease in the process efficiency could be caused by factors different from inhibitor action. Concentration of chromium at which 95% efficiency of the denitrification process was attained depended on the kind of compound used as the electron donor and was different for the individual reactors (Table 1). Further increase in chromium concentration in the influent resulted in a constant decrease in the process efficiency. The concentration of chromium in the effluent was constantly

found to be smaller than in the influent. The reason for this was probably the reduction of chromium by bacterially produced hydrogen sulphide. The possibility of such a reduction has been described by Smillie *et al.* (1981). Chromium Cr^{3+} is practically insoluble. The concentration of soluble Cr^{3+} in equilibrium with $Cr(OH)_3$ is about 10^{-10} mol l^{-1} (Elderfield, 1970). In the water environment considerable quantities of the Cr^{3+} are deposited into the sediments as oxides and hydroxides and possibly adsorbed to particulates as suggested by Curl *et al.* (1965). Inside the reactors a green sludge of chromium hydroxide was found which proved that part of chromium had been reduced to Cr^{3+}.

Table 1. Concentration of chromium which causes a decrease in denitrification efficiency to 95%

Electron donor	Cr^{3+} (mg l^{-1})
Methanol	1.0
Ethanol	3.0
n-Propanol	8.0
n-Butanol	5.0
sec-Butanol	0.5
tert-Butanol	0.0
iso-Butanol	3.5
n-Pentanol	4.0

The results of the inhibition coefficient K_i determination have been presented in Figs 3, 4 and Table 2. From the equations describing the inhibitor

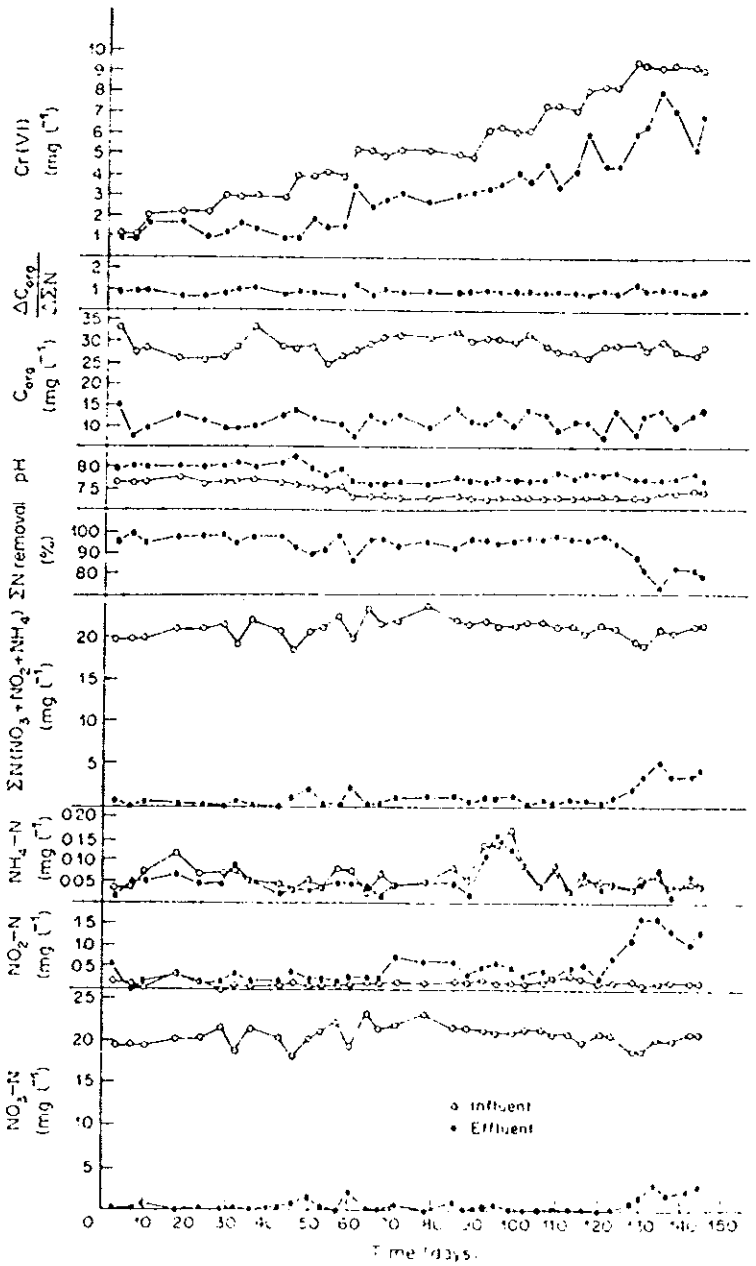

Fig. 2. The performance of the PBR using *n*-propanol as the electron donor.

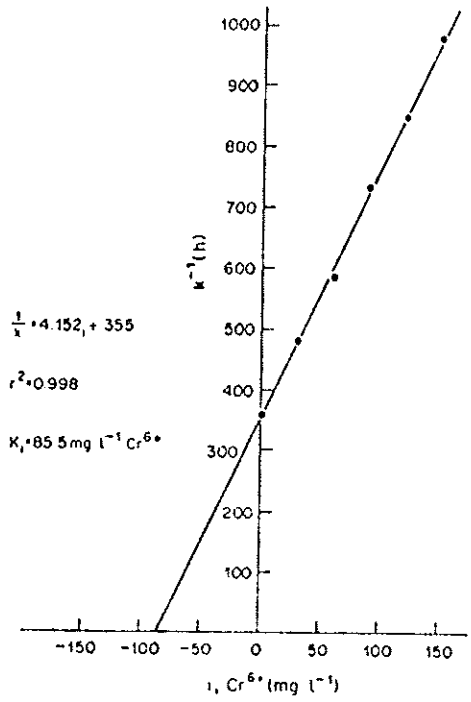

Fig. 3. Determination of the K_i coefficient using methanol as the electron donor.

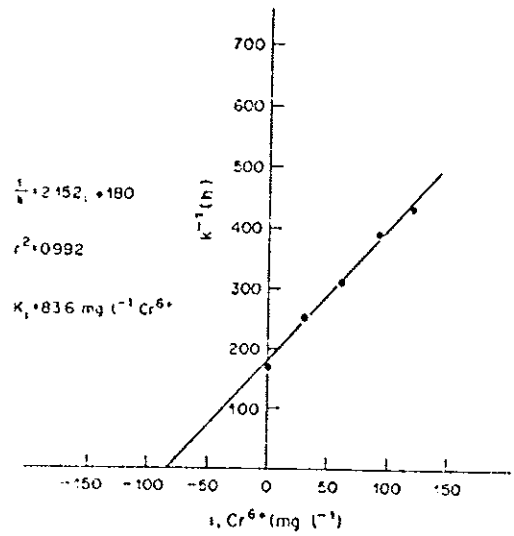

Fig. 4. Determination of the K_i coefficient using *n*-propanol as the electron donor.

influence on the reaction rate the K_i value and the reaction rate in the absence of inhibitor, k_0 have been derived and presented in Table 2. The measured K_i values in all cases were found to be in the relatively narrow range from 81.7 to 86.1 mg l^{-1} Cr^{6+}, which proved that the inhibition coefficient did not depend on the kind of substrate oxidized as the electron donor. The mean value of the K_i coefficient was found to be equal to 84.2 mg l^{-1} Cr^{6+}.

DISCUSSION

The influence of chromium Cr^{6+} on the denitrification process in the PBR was investigated over a long time period. The concentration of chromium was raised step by step and the denitrification efficiency decreased. No adaptation to the presence of chromium during the operating period was observed. From the obtained results it is evident that the tolerance limit of individual reactors to the presence of chromium depends on the kind of carbon substrate oxidized. While the reactor supplied with propanol was operated with full efficiency with the chromium concentration in the influent equal to 8 mg l^{-1} Cr^{6+}, the reactor supplied with sec-butanol did not work properly with the chromium concentration equal to 1 mg l^{-1} Cr^{6+}. On the other hand the inhibition coefficients K_i were almost equal both for propanol and sec-butanol. It was stated on the basis of the results that the concentration of chromium at which the start of inhibitor action is evident depends on the kind of compound utilized as the electron donor. As the criterion of this phenomenon the concentration of chromium which caused a decrease in the denitrification efficiency below 95% was accepted. The concentration of chromium at which the denitrification efficiency decreased to 95% was called "the reactor resistance to inhibition", RRI. The RRI value has been found to be connected with the specific reaction rate of denitrification in the absence of inhibitor, k_0. This dependency is presented in Fig. 5. The values of the specific reaction rates k_0 have been taken from the equations presented in Table 2. Results presented in Fig. 5 proved the linear relationship between the RRI value and the specific reaction rate of denitrification in the absence of

Table 2. The results of the kinetic measurements

Electron donor	Kinetic equation	Inhibition coefficient K_i (mg l^{-1})	Specific reaction rate in the absence of inhibitor $k_0 \cdot 10^{-1}$ (h^{-1})
Methanol	$1/k = 4.152\,i + 355$	85.5	2.82
Ethanol	$1/k = 3.410\,i + 291$	85.3	3.44
n-Propanol	$1/k = 2.152\,i + 180$	83.6	5.56
n-Butanol	$1/k = 2.508\,i + 216$	86.1	4.63
sec-Butanol	$1/k = 4.267\,i + 356$	83.4	2.81
tert-Butanol	$1/k = 5.375\,i + 451$	83.9	2.22
iso-Butanol	$1/k = 2.702\,i + 228$	84.4	4.39
n-Pentanol	$1/k = 2.900\,i + 237$	81.7	4.22

inhibitor:

$$RRI = 2.3\ (k_0 \cdot 10^{-3}) - 5.525. \qquad (11)$$

In the case of the long-term process in the presence of inhibitor the bacterial system is inhibited in two ways: by the inhibition of the reaction rate and by the inhibition of growth. The total reaction velocity depends on the biomass concentration and was measured within the activated sludge as the specific reaction rate in relation to the biomass concentration. The biomass concentration within the PBR was unknown and probably changed during the process course. The observations of the PBR behaviour were restricted to the differences between the chemical compositions of the influent and effluent, the reactor itself considered to be a black-box. The total observed reaction rate could be caused by a small concentration of active microorganisms as well as by a great concentration of microorganisms of small activity. All the investigated PBR had the same detention time. Each PBR was supplied with a different electron donor giving a different reaction rate and probably different growth rate. By hypothetical assumption that the biomass concentration within all the investigated reactors was the same, the reactors supplied with the compounds giving greater reaction rates would need shorter detention time for the process performance. In fact all of them had the same detention time equal to 1 h. When the reactors worked with the efficiency close to 100% some of them which were supplied with the compounds giving the greater reaction rates had reserves and could be operated with the detention time shorter with the same results of 100% denitrification. Introduction of inhibitor caused a decrease in reaction rate. The same decreases in reaction rates were detected faster in the case of reactors supplied with the compounds giving the smaller reaction rate. In the other reactors the "reserve" in reaction rate has been reduced with no influence on the obtained results. The bacterial systems in all of the reactors were influenced by chromium to the same degree. Every one of the reactors had anyhow a different "reserve" in reaction rate which could be reduced by inhibitor action with no influence on the observed process efficiency. Every one showed a different reactor resistance to inhibition because every one was supplied with a compound giving a different reaction rate. This resulted in the linear relationship between the RRI value and the reaction rate as it was presented in Fig. 5.

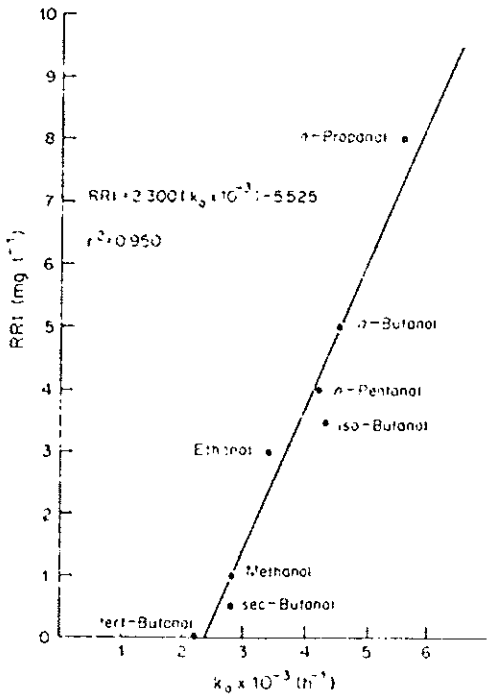

Fig. 5. Reactor resistance to inhibition, RRI vs the denitrification rate in the absence of inhibitor, k_0.

The expression of "reactor resistance to inhibition" should be restricted to description of the reactor behaviour, not for bacterial system itself. In this way the literature data expressed in form of "the bacterial system, for example activated sludge, tolerates the defined concentration of cyanide, phenol and the like" leads to misunderstanding because the measured value of the tolerance limit depends on the reactor operation conditions. No adaptation of bacterial system to the presence of chromium was observed during the described investigations. Anyhow, even in the case when some adaptation mechanisms are involved the consideration should be given to the reactor resistance to inhibition which came from the conditions of the measurement, not from the bacterial system properties. In this work the measurement of the inhibition coefficient K_i was proposed. The method has been adapted from pure enzymology and seems to be a good tool for inhibitor action assessment.

According to the previously described Dixon's method a linear relationship is obtaining on plotting $1/V$ against i. In the case of the described experiments the expression of $1/V$ has been replaced by $1/k$ which did not change the K_i value but was more convenient for the further considerations. The K_i value has been found to be independent of the kind of compound used as the electron donor in the respiration chain. It was proved in other work by the author (Lewandowski *et al.*, 1985) that the K_i value is independent of the biomass concentration as well. To compare the behaviour of two microbial systems in the presence of the same inhibitor but supplied with different electron donors the two theoretical lines have been drawn in Fig. 6. From this sketch it is seen that for the same inhibitor the value of K_i is the same, only the slopes of the lines are different. That means that the same increase in the inhibitor concentration causes different changes in the reaction rate, depending on the substrate utilized. The slope of this line has been called "the specific inhibition sensitivity" SIS

$$SIS = \frac{1}{k_0 \cdot K_i} \qquad (12)$$

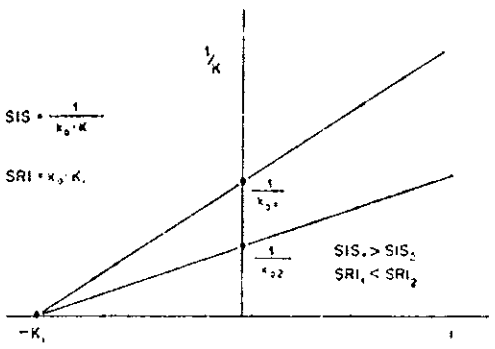

Fig. 6. The specific inhibition sensitivity, SIS and the specific resistance to inhibition, SRI.

where:

k_0 = the specific reaction rate in the absence of the inhibitor.

The SIS value is characteristic for the compound utilized as the electron donor by the bacterial system. When the SIS value increase the sensitivity of the bacterial system to the presence of inhibitor increases as well. The same increase in the inhibitor concentration results in greater decrease in reaction rate when the compound giving the greater inhibition sensitivity is used as the electron donor. The inverse of the specific inhibition sensitivity has been called "the specific resistance to inhibition", SRI

$$SRI = k_0 \cdot K_i \tag{13}$$

From the data presented in Table 2 results the mean value of K_i equal to 84.2 mg l^{-1} Cr^{6+}. By multiplying this value by the values of the specific reaction rates in the absence of inhibitor, k_0, the values of SRI presented in Table 3 have been obtained for the individual compounds used in the experiment. The relationship between the specific resistance to inhibition SRI and the reactor resistance to inhibition RRI is the same in shape as the one presented in Fig. 5. The values on the x-axis have been multiplied by the constant K_i value. The resulting relationship has the form of:

$$RRI = 27.316\ SRI - 5.527 \tag{14}$$

with $r^2 = 0.949$

The effect of the biomass concentration on the reactor resistance to inhibition has not been investigated in this work because of the kind of reactor used in the experiments. According to the presented concept the toxicity effect should be inversely proportional to the biomass concentration within the reactor because the total resistance to inhibition is equal:

$$RI = SRI \cdot x \tag{15}$$

where:

RI = resistance to inhibition,
x = biomass concentration.

Some literature data concerned with the toxic effect of heavy metals can be utilized to support this thesis. In the work by Lamb and Tollefson (1973) the toxic effect of cupric ions on biological oxidation was investigated. The experiments were carried out in a laboratory scale continuous flow activated sludge reactor and the toxic effect was measured as the changes in glucose conversion velocity within the reactor versus the concentration of the cupric ions. The authors come to the conclusion that: "the toxicity decreased markedly with increased suspended solids concentration: an 80% decrease in conversion at 210 ppm suspended solids was reduced to negligible quantity 3% by increasing the suspended solids to 4000 ppm". In that work the method for reducing the toxic effect by increasing the solids concentration was suggested. The data and conclusions fit well the concept of the reactor resistance to inhibition. Anyhow, the explanation of the authors that "the greater proportion of dead cells at higher suspended cell concentrations may interact with the toxic metal ions to reduce their toxicity" seems doubtful.

If we presume that under the same conditions the same concentration of inhibitor inhibits constantly the same concentration of biomass we can write, by i = const:

$$x \cdot I = \gamma \tag{16}$$

where:

x = biomass concentration,
I = inhibition rate,
γ = constant, concentration of the biomass being inhibited.

In the case of the results obtained by Lamb and Tollefson by the biomass concentration equal to 210 ppm:

$x = 0.21$ g l^{-1} and $I = 0.80$ gives $\gamma = 0.16$ g l^{-1}.

When the biomass concentration increases to 4000 ppm:

$x = 4.00$ g l^{-1} and $\gamma = 0.16$ g l^{-1} gives $I = 0.04$.

The predicted value of inhibition rate 0.04 is close to the one measured by the authors, equal to 0.03.

Another conclusion of the authors that "toxic effects on the operation of lagoons could possibly be far greater than in an activated sludge system because of the lower suspended solids concentration" seems to be a particular case from the point of view of the

Table 3. The values of the SRI for the compounds used as electron donors

Electron donor	SRI (mg l^{-1} h^{-1})
Methanol	0.237
Ethanol	0.290
n-Propanol	0.468
n-Butanol	0.390
sec-Butanol	0.237
tert-Butanol	0.187
iso-Butanol	0.370
n-Pentanol	0.355

reactor resistance to inhibition concept. The RRI value depends not only on the biomass concentration but also on the detention time, the kind of substrate oxidized and the temperature as well. So that in the case of lagoons the disadvantageous effect of the small biomass concentration can be reduced by prolonged detention times.

CONCLUSIONS

(1) Despite the long operation time of the packed bed reactors no adaptation of bacterial systems to the presence of chromium Cr^{6+} was observed.

(2) The packed bed reactors supplied with various organic compounds as the electron donors for microorganisms and operated under identical conditions showed different reactor resistance to inhibition, RRI.

(3) The reactor resistance to inhibition defined as the initial concentration of inhibitor at which the obtained results in terms of the process efficiency are deteriorated may be explained. The reason for this is the difference between the time needed for the process performance and the detention time. The excess detention time over that required leads to existence of "reserve" in reaction rate which may be reduced by inhibitor action without deterioration of the obtained results.

(4) The biological reactors can be designed with the determined reactor resistance to inhibition RRI, according to the expected inhibitor concentration.

(5) The literature data on the tolerance limits of bacterial systems to the presence of selected inhibitors are relative in character because the measured "tolerance limit" depends on the measurement conditions.

(6) The inhibition coefficient K_i measurement seems to be a good tool for the inhibitor action assessment.

(7) The introduced parameters of specific inhibition sensitivity and specific resistance to inhibition seem to be convenient to characterize the bacterial system being inhibited.

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