European Chemicals Bureau

Institute for Health and Consumer Protection

European Chemicals Bureau

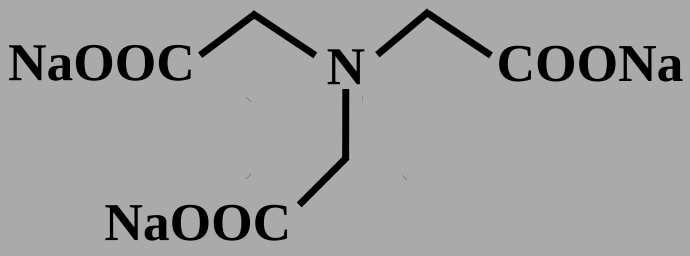
Existing Substances

European Union Risk Assessment Report

CAS No: 5064-31-3

EINECS No: 225-768-6

trisodium nitrilotriacetate



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European Union Risk Assessment Report

TRISODIUM NITRILOTRIACETATE

Part I – Environment

CAS No: 5064-31-3

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RISK ASSESSMENT

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Part I – Environment

CAS No: 5064-31-3

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RISK ASSESSMENT

Final Report, 2005

Germany

The risk assessment of trisodium nitrilotriacetate has been prepared by Germany on behalf of the European Union.

Contact point:

Bundesanstalt für Arbeitsschutz und Arbeitsmedizin
Anmeldestelle Chemikaliengesetz (BAuA)
(Federal Institute for Occupational Safety and Health Notification Unit)
Friedrich-Henkel-Weg 1-25
44149 Dortmund
Germany

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Review of report by MS Technical Experts finalised:	**2001**
Final report:	**2005**

Foreword

We are pleased to present this Risk Assessment Report which is the result of in-depth work carried out by experts in one Member State, working in co-operation with their counterparts in the other Member States, the Commission Services, Industry and public interest groups.

The Risk Assessment was carried out in accordance with Council Regulation (EEC) 793/93[1] on the evaluation and control of the risks of "existing" substances. "Existing" substances are chemical substances in use within the European Community before September 1981 and listed in the European Inventory of Existing Commercial Chemical Substances. Regulation 793/93 provides a systematic framework for the evaluation of the risks to human health and the environment of these substances if they are produced or imported into the Community in volumes above 10 t/year.

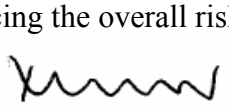
There are four overall stages in the Regulation for reducing the risks: data collection, priority setting, risk assessment and risk reduction. Data provided by Industry are used by Member States and the Commission services to determine the priority of the substances which need to be assessed. For each substance on a priority list, a Member State volunteers to act as "Rapporteur", undertaking the in-depth Risk Assessment and recommending a strategy to limit the risks of exposure to the substance, if necessary.

The methods for carrying out an in-depth Risk Assessment at Community level are laid down in Commission Regulation (EC) 1488/94[2], which is supported by a technical guidance document[3]. Normally, the "Rapporteur" and individual companies producing, importing and/or using the chemicals work closely together to develop a draft Risk Assessment Report, which is then presented at a Meeting of Member State technical experts for endorsement. The Risk Assessment Report is then peer-reviewed by the Scientific Committee on Health and Environmental Risks (SCHER) which gives its opinion to the European Commission on the quality of the risk assessment.

If a Risk Assessment Report concludes that measures to reduce the risks of exposure to the substances are needed, beyond any measures which may already be in place, the next step in the process is for the "Rapporteur" to develop a proposal for a strategy to limit those risks.

The Risk Assessment Report is also presented to the Organisation for Economic Co-operation and Development as a contribution to the Chapter 19, Agenda 21 goals for evaluating chemicals, agreed at the United Nations Conference on Environment and Development, held in Rio de Janeiro in 1992.

This Risk Assessment improves our knowledge about the risks to human health and the environment from exposure to chemicals. We hope you will agree that the results of this in-depth study and intensive co-operation will make a worthwhile contribution to the Community objective of reducing the overall risks from exposure to chemicals



Roland Schenkel
Acting Director-General
DG Joint Research Centre

Catherine Day
Director-General
DG Environment

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[1] O.J. No L 084, 05/04/1993 p.0001 – 0075

[2] O.J. No L 161, 29/06/1994 p. 0003 – 0011

[3] Technical Guidance Document, Part I – V, ISBN 92-827-801 [1234]

0 OVERALL RESULTS OF THE RISK ASSESSMENT

CAS No:	5064-31-3
EINECS No:	225-768-6
IUPAC Name:	Trisodium nitrilotriacetate

Environment

Conclusion (ii) There is at present no need for further information and/or testing and for risk reduction measures beyond those which are being applied already.

In the present risk assessment production and use of Na_3NTA are examined. For all life-cycle steps, the PEC/PNEC ratios are below 1. Therefore, a risk for the environment is not expected.

CONTENTS

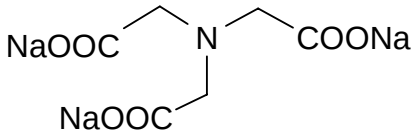
TABLES

Euses Calculations for regional exposure can be viewed as part of the report at the website of the European Chemicals Bureau: http://ecb.jrc.it

1 GENERAL SUBSTANCE INFORMATION

1.1 IDENTIFICATION OF THE SUBSTANCE

CAS No:	5064-31-3
EINECS No:	225-768-6
IUPAC Name:	Trisodium nitrilotriacetate
Synonyms:	Nitrilotriacetic acid trisodium salt, NTA trisodium salt
CA Index name:	Glycine, N,N-bis(carboxymethyl)-, trisodium salt
Empirical formula:	$C_6H_6NNa_3O_6$
Molecular weight:	257.1 g/mol
Structural formula:	



1.2 PURITY/IMPURITIES, ADDITIVES

Purity:	≥ 92% w/w
Impurity:	< 7% water < 3 sodium glycolate < 2% disodium iminodi(acetate) < 2 sodium hydroxide < 1.5% methanamine < 1 sodium formate
Additives:	none

1.3 PHYSICO-CHEMICAL PROPERTIES

Trisodium nitrilotriacetate (Na_3NTA) is a colourless crystalline powder at room temperature and normal pressure. Data on the physical and chemical properties are given in **Table 1.1**.

Table 1.1 Physico-chemical properties

Parameter	Value	Reference
Melting point	410°C with decomposition above 200°C	BASF, 1996
Boiling point	not applicable	
Relative density	1.77 at 20°C 1)	BASF, 1997
Vapour pressure	not determined	no test conducted because of structural reasons
Surface tension	not determined	no test conducted because of structural reasons
Water solubility	about 640 g/l at 20°C	Ullmann, 1991
Partition coefficient	-2.62 (calculated) 2)	BASF, 1989
Flash point	not determined	substance is a solid
Flammability	not highly flammable 3)	BASF, 1998
Ignition temperature	no self ignition up to decomposition (200°C)	Chemsafe, 1997
Explosive properties	not explosive	no test conducted because of structural reasons
Oxidising properties	no oxidising properties	no test conducted because of structural reasons

1) Relative density: pycnometer method
2) Partition coefficient: The logPow was calculated according to the Rekkermethod with the computer program Pro-logP (version 2.0).
3) Flammability: According to guideline 92/69/EEC test method A.10 (determination of the flammability for solids), the substance did not propagate combustion. The tests according to A.12 (determination of the flammability in contact with water) and A.13 (determination of pyrophoric properties) were not conducted. Due to the properties and the handling of the substance it has not to be assumed that flammable gases formate in contact with water or the substance has pyrophoric properties.

1.4 CLASSIFICATION

Classification according to Annex I

Class of danger: none
R-Phrase: none

The classification of NTA is not included in Annex I to Derective 67/548/EEC.

Proposed classification (only environmental part)

For the classification of biodegradation, the available laboratory tests with uncomplexed NTA should not be used, because biodegradation of this chelator is strongly dependent on the metal speciation. Studies on the degradation in biological treatment plants as well as degradation tests conducted in river water reveal that the degradation properties of NTA (its metal complexes respectively) are comparable with a readily degradable substance. In addition the substance has no bioaccumulation potential.

In tests on the acute toxicity on fish and daphnia effects were only observed when NTA was present in over-stoichiometric concentrations compared to the content of metal ions. The lowest LC_{50} values were 98 mg/l for both trophic levels.

Results of algae growth inhibition tests have to be interpreted carefully, because the observed effects are mainly cause by nutrient deficiency, which is an artefact and not relevant for the environment. Tests with increased concentrations of nutrient metals (where nutrient deficiency is

suppressed) reveal that intrinsic toxicity of NTA is expected only at concentrations far above 10 mg/l.

Considering all results, a classification of Na_3NTA for the environment is not recommended.

2 GENERAL INFORMATION ON EXPOSURE

NTA is produced and used as sodium salt (Na_3NTA) or as acid (H_3NTA). During the use and, after release into the environment, complexes with metal ions are formed. The environmental exposure of all NTA complex species is overlapping. In the scientific literature dealing with the environmental fate and toxic effects on organisms, amounts and concentrations are mostly referred as Na_3NTA. Thus, for the environmental risk assessment (Sections 2 and 3) all production and use volumes are given as Na_3NTA equivalents.

2.1 PRODUCTION

2.1.1 Production processes

Today the original synthesis of NTA from ammonia and chloroacetic acid has only historical significance. The oxidation of triethanolamine is likewise of no industrial importance. The one-stage alkaline and two-stage acid processes now in use are based on the cyanomethylation of ammonia (or ammonium sulfate) with formaldehyde and sodium cyanide (or hydrogen cyanide).

The alkaline process was long the established method for NTA production. Trisodium nitrilotriacetate is synthesised as follows:

$$NH_3 + 3\ HCHO + 3\ NaCN \rightarrow N(CH_2CN)_3 + 3\ NaOH$$

$$N(CH_2CN)_3 + 3\ NaOH + 3\ H_2O \rightarrow N(CH_2COONa)_3 + 3\ NH_3$$

The reaction can be carried out batch wise or continuously, but the continuous process is more economical. The resulting solution is sold directly as a 40-wt% solution, or used in the production of Na_3NTA in powder form, or acidified to pH 1 - 2 to yield the acid (H_3NTA).

Acid Process: The significant yield of by-products in the alkaline process has led in recent years to the construction of plants based on the acid process, which features much lower by-product levels. The acid process is associated with stringent safety requirements due to the use of hydrogen cyanide; corrosion can also be a problem. In the first stage, ammonia is reacted with formaldehyde to give hexamethylenetetramine, which is then reacted with hydrogen cyanide in sulfuric acid solution to yield triscyanomethylamine. The solid triscyanomethylamine is sparingly soluble in the acidic solution and is filtered off, washed, and saponified with NaOH to give Na_3NTA. The resulting solution has a far lower by-product content than the solution from the alkaline method. It is also sold as 40% product or used in the production of Na_3NTA or H_3NTA (see above).

2.1.2 Production volumes

The following companies are producer and/or importers of NTA:

- Akzo Nobel Chemicals B.V., Herkenbosch (NL)
- Akzo Nobel Chemicals B.V., Kvantorp (SWE)
- BASF AG, Ludwigshafen (GER)
- Dow Europe S.A., Seal Sands (UK)
- Solutia Europe S.A. (BEL)

According to the data supplied by the producers and importers for this report, 36,090 tonnes/annum (calculated as Na_3NTA) are produced, 6,040 tonnes/annum are imported and 10,090 tonnes/annum exported outside of the EU, thus 32,040 tonnes/annum are consumed within Europe. According to CEFIC (2001), 26,642 tonnes/annum were marketed in 2000, the difference to the producers data might be explained by exports or imports of NTA containing formulations.

The NTA amounts (calculated as Na_3NTA) marketed in the Western European countries are given in the **Table 2.1**. The figures are derived from sales information of the producers. A direct correlation to the consumption volume is therefore not precise; however the figures may be regarded as an approximation for the European consumption. Imports and exports of Na_3NTA containing formulations are not considered.

Table 2.1 Na_3NTA Sales in European Countries (CEFIC, 2000)

Country	Sales (tonnes) in 1999
Germany	3,396
Belgium / Luxembourg	1,814
The Netherlands	2,055
France	1,838
Italy	825
UK	7,274
Ireland / Denmark	561
Spain / Portugal / Greece	5,108
Finland	Not published
Norway	0
Sweden	1,734
Austria	Not published
Switzerland	631
Total West. Europe	26,756

2.2 USE

NTA is an aminocarboxylic acid with three functional groups which donate electrons. These enable it to participate in complexation reactions. The most important property of NTA is to form water-soluble complexes with multivalent metal ions over a wide pH range.

NTA and its sodium salt are used to soften water and to remove traces of alkaline earth and heavy metals. They are often included in detergent and cleaner formulations for household or industrial use.

The application volumes (calculated as Na_3NTA) were (CEFIC, 2000; CEFIC 2001):

Table 2.2 Use pattern of Na_3NTA (CEFIC, 2000; CEFIC 2001)

	IC/UC *	Germany (1999)	Western Europe (2000)
Marketed amount		3,396 tonnes	26,642 tonnes
Textile cleaning, household and industrial	5/11 13/11	238 tonnes (7%)	973 (3.7%)
Cleaning agents	6/9	2,207 (65%)	17,905 (67.2%)
Others	15/0	951 (28%)	7,764 (29.1%)

* Industrial category / use category

3 ENVIRONMENT

3.1 ENVIRONMENTAL EXPOSURE

3.1.1 Environmental releases

Production

During production, releases occur via waste water into the hydrosphere. According to the data submitted by the producers, the total yearly releases into the hydrosphere are 24.1 tonnes/annum (see Section 3.1.3.1).

Use

During the use as complexing agent, the major amount of the applied NTA is released into the waste water. The emission situation for the individual uses is presented in Section 3.1.5.2.

Frequently the question is raised in the literature whether NTA can cause hazardous effects due to its property to keep heavy metal ions in the water phase of rivers. The interaction of NTA with metal ions is elaborated in Section 3.1.3.5.

3.1.2 Environmental fate

3.1.2.1 Degradation

3.1.2.1.1 Biodegradation

Laboratory biodegradation tests

A series of laboratory degradation tests is available for NTA (see **Table 3.1**). In most cases the acid or the Na-salt was added and not the complexed NTA. However, the test media generally contain, beside trace metals, calcium and/or magnesium ions in over-stoichiometric amounts, the respective complexes are formed thus being the active test substances.

The test results indicate that the Ca/Mg complexes are readily removed. For instance, in a Modified OECD Screening Test conducted according to OECD guideline 301 E, NTA was found to be readily biodegradable. Meeting the 10 days time window criterion, the substance (initial concentration 50 mg/l) was completely degraded within 14 days, as measured by DOC. The inoculum used was taken from river water treatment plant. The lag phase until degradation started was 5 days (BASF, 1983b).

Mechanism of Degradation

Several bacteria strains capable of growth with NTA were isolated from wastewater, soil and sediment. An aerobic *Chelatobacter* strain a monooxigenase is responsible for the initial oxidation, leading to iminodiacetate (IDA) and glyoxylate. IDA is subsequently oxidised to glycine and glyoxylate by a membrane-bound dehydrogenase. Denitrifying bacteria contain a

dehydrogenase/nitrate reductase complex, catalysing the formation of IDA, glyoxylate and nitrite from NTA and nitrate (Egli, 1992, 1994).

The reaction pathway supports the result of the standard tests that NTA is completely mineralised after primary degradation. Accumulation of a stable metabolite is not expected.

Table 3.1 Results of laboratory biodegradation tests

Type	Method	Duration (day)	Inoculum [1)]	Na_3NTA conc. (mg/l)	Degradation	Lag phase (day)	Reference
Modified OECD Screening Test	OECD 301 E	14	River water	70	100%	5	BASF (1983b)
Modified OECD Screening Test	OECD 301 E	14	Industrial WWTP effluent	70	100%	5-11	BASF (1983b)
Modified OECD Screening Test	OECD 301 E	7	Adapted AS	70	100%	1	BASF (1983c)
Modified OECD Screening Test	OECD 301 E	12	Adapted AS	140	75-90%	2-5	BASF (1983c)
Sturm Test	CO_2 evol.	9	Effluent from stand test	10/20	100%	-	BASF (1983d)
Manometric Respirometry Test	OECD 301 F	28	Industrial AS	250-360	92%	16	Strotmann et al. (1995)
Combined CO_2/DOC Test	Other	28	Industrial AS	140	>95% DOC 91% CO_2	2 (DOC) 5 (CO_2)	Strotmann et al. (1995)
Modified Zahn-Wellens Test	OECD 302 B	28	Industrial AS	1,400	96%	7	BASF (1983a)
Die-away Test	Other	23	Municipal AS	210	100%	14	Takahashi et al. (1997)

1) AS: Activated Sludge

Influence of heavy metals on biodegradation

It was demonstrated by several investigations that biodegradation of NTA is influenced by the metal speciation.

Bench scale batch activated sludge experiments were conducted with various NTA-metal complexes with initial concentrations (as H_3NTA) between 8 and 16 mg/l (Shannon et al., 1978). The inoculum was activated sludge from a treatment plant; the test was run at 5°C. Biodegradation could be approximated by a first-order reaction. The degradation rates are presented in the table below. Oxygen uptake values observed during the experiments indicated that, with the exception of mercury, there were no inhibitory effects by the metals.

Bolton et al. (1996) used the NTA-degrading bacterium *Chelatobacter heintzii* in batch experiments with 5.23 µM (=1.3 mg/l Na_3NTA) complex solutions. Biodegradation followed first-order kinetics. Glucose degradation experiments in the presence of NTA-metal complexes revealed that metal toxicity was not a factor limiting NTA degradation. To test the hypothesis that the rate of NTA degradation would decrease as the complex stability constant increases, metals with a range of NTA stability constants were chosen. Calculations of the aqueous speciation using the computer program MINTEQ demonstrate that the added complexes were the dominant species in all experimental solutions. No relationship was found between the magnitude of the thermodynamical stability constant for the complexes and the first-order rate constant. E.g. Fe(III) which forms the thermodynamical most stable complex is rapidly degraded. Instead of this, a correlation between degradation constant and the liability (i.e. the reaction rate of complex dissociation) was found. The authors assume that the Mg-complex is the exclusive substrate for NTA degradation, its formation is related to the relative rates of $HNTA^{2-}$ formation from the metal complex dissociation.

Table 3.2 Influence of metal ions on biodegradation

NTA complex with	Degradation constant (h^{-1})		Stability constant $\log K_{st}$	Dissociation constant ($M^{-1}\ s^{-1}$)
	Shannon et al. (1978)	Bolton et al. (1996)	Bolton et al. (1996)	Bolton et al. (1996)
uncomplexed		0.672	0	
Pb^{2+}	0.072			$4.4 \cdot 10^6$
Ca^{2+}	0.067			
Co^{2+}		0.260	11.7	$0.8\text{-}8 \cdot 10^5$
Fe^{3+}	0.066	0.209	17.8	
Zn^{2+}	0.051	0.190	11.9	$5.1 \cdot 10^5$
Al^{3+}	0.057	0.124	13.7	
Cr^{3+}	0.040			
Cu^{2+}	0.025	0.114	14.2	$1.1 \cdot 10^5$
Cd^{2+}	0.008			$2.1 \cdot 10^5$
Ni^{2+}	0.005	0.063	12.8	7.5
Hg^{2+}	0			

Madsen and Alexander (1985) studied the degradation using sewage organisms taken from raw sewage of a municipal treatment plant. Ca-NTA was mineralised at concentrations of 1, 10, and 100 µg/l. As the concentration of Ca-NTA increased, the rate of breakdown increased. No

mineralisation of Al, Mg, H, or Fe-NTA was detected with the same concentrations after an incubation of 585 hours.

The available investigations demonstrate that results received from tests with uncomplexed NTA and with Ca- or Mg-complexes should interpreted with care. Both in waste water and the hydrosphere, heavy metal ions are present which may inhibit degradation. For the environmental exposure assessment, the degradation properties of the metal complexes instead of the uncomplexed agent have to be taken into account.

There are contradicting results about the degradable metal species. E.g. according to Bolton et al. (1996) the uncomplexed NTA is most rapidly degraded, Madsen and Alexander (1985) found that only the Ca-complex is mineralised. The contradicting results may be caused by specific degradation mechanisms in individual inoculi. There is an agreement that complexes with several heavy metals inhibit NTA degradation. Heavy metals are widely spread; considerable fractions of NTA may be bound to these metals thus influencing the agent's removal in treatment plants and in environmental compartments.

However, non-degradable complex species can be transformed into degradable compounds by metal exchange reactions. As in sewage and the hydrosphere always a mixture of complexes is present, a prediction of the degradation rates from standard laboratory tests is not possible.

Elimination in treatment plants

An extended study was conducted in the Netherlands using a pilot plant of some 3,000 population equivalents capacity. The sewage (municipal with a high industrial fraction) was spiked with H_3NTA up to 40 mg/l in 3 steps. The observed adaption time was 0 – 9 days dependent on the concentrations. With a continuous NTA load, removal rates of 93.0 – 98.9% (an average of 96.2%) were observed after adaption. The removal decreased when the plant was overloaded with organic waste or when the temperature fell below 7°C (Pöpel et al., 1984).

In 1984, NTA was detected in influent and effluent of 2 German municipal biological treatment plants. The concentrations were in the range of 196 – 562 µg/l in the influent and 6.6 – 23.1 µg/l in the effluent, the calculated removal is 93.5% – 98.2% (average 96.2%). The measurements were carried out in winter with effluent temperatures in the range of 11 – 13°C. The hydraulic retention time was 5 – 10 hours. For a further plant a removal of 90% is reported (Hansen, 1986).

In a Swiss municipal plant, influent concentrations of 300 – 1,500 µg/l (diurnal variation) were detected. In both seasons, NTA was degraded to 97%. The average sludge age was 3.6 days in winter and 4.8 days in summer. The influent concentrations increased by a factor of 4 between 1984 and 1987, whereas the effluent concentrations rose only by a factor of 1.5 (Alder et al., 1990).

For a Canadian municipal activated sludge plant, a removal of >95% in summer and less than 50% in winter is reported. The winter temperature in the plant is not reported, in the receiving creek the range was 0.5 – 3°C (Shannon et al., 1974).

The influence of temperature on the removal rate was investigated by Stephenson et al. (1983b) using activated sludge pilot plants. With 15 mg/l H_3NTA and typically occurring heavy metal concentrations, the removal rate (98%) was essentially unchanged when the temperature decreased from 17.5 to 10°C, further temperature reduction to 6°C caused a decrease in removal to 85%. With higher heavy metal concentrations (typically for industrial sewage), the removal was 95.2% (17.5 °C), 92.9% (10°C), and 79% (6°C).

In a pilot trickling filter plant, NTA was degraded incompletely. The removal of 20 mg/l in the influent was 70 - 80% above 10°C and 60 - 70% below 10°C in winter. With 40 mg/l in the influent, a removal of 67 – 82% above 10°C and <50 – 67% was observed. In a pilot oxidation ditch, the removal was <75% below 10°C (Heide, 1983).

The anaerobic degradation in sewage sludge was studies by Stephenson et al. (1983a). H_3NTA was added in concentrations of 10 – 30 mg/l to four laboratory scale anaerobic digesters treating mixed primary sludge and incubated at 35°C. After periods of adaption between 4 and 16 days, NTA was removed within some days. When NTA addition was interrupted, a memory effect was observed over periods up to 30 days.

The available studies reveal that NTA is removed in municipal treatment plants with rates generally above 95% under normal operation conditions. Contradicting results were obtained for measurements in the winter season: while in several cases no difference between summer and winter was observed, other studies show considerable decrease at low temperatures. Removal is disturbed when the temperature falls below a threshold of about 7°C. For the exposure calculations in this assessment, a removal of 95% is used; having in mind that in harsh winters the NTA releases can be increased.

Biodegradation in natural waters

NTA degradation was studied in a Canadian creek receiving municipal sewage from a biological treatment plant. The study was designed to determine the influence of temperature under summer and winter extremes. NTA was monitored at 6 sampling stations downstream. During the summer period, the NTA concentration was consistently less than the detection limit of 10 µg/l. In winter when the stream temperatures ranged from 0.5 to 3°C and NTA removal through the treatment plant was less than 50%, the downstream concentrations averaged to 106 µg/l. Considering the flow conditions, there was still evidence of biodegradation although it was considerably reduced. To determine the reaction rates, laboratory studies were carried out: in a 50 l tank containing creek water and a 10 cm sediment layer, primary degradation of 200-250 µg/l H_3NTA was determined at temperatures between 2 and 18°C. The degradation could be approximated by a first-order reaction with a rate constant ranging from 0.036 h^{-1} ($t_{1/2}$ = 19 hours) at 18°C to 0.006 h^{-1} ($t_{1/2}$ = 115 hours) at 2°C (Shannon et al., 1974).

Acclimation to and biodegradation of NTA was studied at trace concentrations in several river waters. Uniformly labelled ^{14}C-Na_3NTA was added in various concentrations (1 – 1,000 µg/l) at 14 or 24°C. In river waters not previously exposed to NTA, acclimation and degradation were observed at the lowest concentration tested, which indicates that no threshold effect is expected. Degradation was preceded by a 5 – 10 day lag phase, and followed first-order kinetics after acclimation, the mineralisation half-life ranged from 7 to 138 h at initial concentrations of 50 and 5 µg/l, respectively. In water samples where prior NTA exposure had already occurred no acclimation was required and degradation ($t_{1/2}$ = 7 – 17 hours) was less variable than in unexposed rivers (Larson and Davidson, 1982).

The effect of temperature on NTA degradation was investigated by Kari (1994). In the Swiss river Glatt receiving a high load of waste water, over a flow distance of 22 km about 90% of NTA was eliminated in summer and 65% in winter.

The biodegradation of NTA in the estuarine environment was examined by using a laboratory estuarine simulation (Hunter et al., 1986). Series of 5 reaction vessels were arranged in a stepped sequence, and a saline gradient (1.11 – 17.5%) was achieved. The retention time for each vessel was between 5.6 and 8.3 hours. Natural microbial populations from freshwater and marine

sources were used as inoculum. Na_3NTA concentrations (0.9 – 1.02 µg/l) were removed to more than 95% in a salinity up to 8.77% after an acclimation time of 11 – 15 days with higher salinities, the removal decreased to minimum 48%. Na_3NTA concentrations of 6.95 mg/l (which is above the expected environmental level) were not removed in salinities above 20%. The authors conclude that bacteria are unable to acclimate to NTA at moderate salinities.

The effects of salinity and DOC on the kinetics of biodegradation of NTA were studied in a Canadian river estuary with a prior history of NTA exposure (Larson and Ventullo, 1986). Water samples were collected in distances of 300 – 4,600 m from the outfall of a primary treatment facility. $U^{14}C$-NTA was added in a range of 10 – 100 µg/l, and $^{14}CO_2$, ^{14}C activity in biomass, and ^{14}C remaining in solution was measured. Degradation occurred immediately with no apparent lag phase, indicating that the microbial communities were adapted as a result of prior NTA exposure. There was no consistent effect of salinity (range 4 – 19%) or DOC (range 2-12 mg/l) levels on NTA degradation rates. Degradation followed first-order kinetics; the estimated mineralisation half-life was about 2 days.

The biodegradation of $U^{14}C$-NTA at concentrations of 1 – 1,000 µg/l in German estuarine water was determined by Hales and Ernst (1991). The final extent of degradation was lower at high salinity and low NTA concentrations. The shapes of the biodegradation curves was biphasial, degradation occurred in two stages in cases where microbial growth was required for complete degradation. From the initial phase, occurring under all conditions of temperature and salinity, half-life of 4 – 29 days at 12.2°C were determined. The second stage was fitted by non-linear regression with the integrated Monod growth model: the 1,000 µg/l concentration gave μ_{max} values in the range of 0.9 – 1.7 day^{-1}.

The available studies demonstrate that NTA is biodegraded in freshwaters with half-life in the range of several hours to some days, after an acclimation period in the range of days to weeks. Degradation is considerably influenced by temperature. For the following exposure calculations, a half-life of 5 days is used.

Contradictory results have been reported on its degradation in marine or estuary waters. Whereas Hunter et al. (1986) found that marine bacteria were unable to acclimate; rapid degradation ($t_{1/2}$ = 2 days) was found for Canadian rivers with a prior history of NTA release (Hales and Ernst, 1991). Without acclimation increased half-lives up to several weeks are expected. The latter results are not considered in the assessment, as in the time frame of several days in estuaries dilution processes are predominating.

Biodegradation in sediments

Adaptation of bacterial activity for the primary degradation of NTA was studied using natural sediment samples and an NTA-degrading bacterium (*Pseudomonas sp.*). Sediment samples collected from a river loaded with persistent NTA levels degraded NTA with a half-life of about 1 day at a temperature of 30°C. The reaction rate with the pure bacteria strain associated with sand was approximately one order of magnitude slower. With sediment from a less contaminated control site a 5 day lag preceded an abrupt increase in NTA degradation (McFeters et al., 1990).

The only available study demonstrated a NTA degradation half-life of 1 day. However, the temperature (30°C) is far above typical environmental conditions, thus the real degradation rate is possibly lower. For the exposure calculations, for the aerobic sediment layer the same half-life (9 days) than for soils is used.

No studies with anaerobic sediments are available. NTA was found to be degradable in anaerobic sludge (Stephenson et al., 1983a) and with soil organisms under anaerobic conditions (Tabatabai

and Bremmer, 1975; cited below). The reaction was slower than for the aerobic degradation. For degradation in the anaerobic sediment layer, a roughly estimated half-life of 20 days is used.

Biodegradation in soil

The primary degradation of NTA in 5 soils was studied by performing analyses for NTA and inorganic N. After 5 days incubation at 30°C, 12 - 45% of the Na_3NTA (initially 200 mg/kg) was recovered under aerobic and 20 - 65% under anaerobic conditions After incubation under anaerobic conditions the NTA-nitrogen was in the form of NH_4^+, while under aerobic conditions NO_3^- was formed (Tabatabai and Bremmer, 1975).

Mineralisation of $U^{14}C$-NTA was measured in batch test systems containing groundwater or groundwater plus subsurface soil material (GWSS slurries) at 20°C. In groundwater, with an initial concentration of 50 µg/l, a half-life of 54 hours was determined under aerobic conditions. About 80% of the radioactivity was evolved as $^{14}CO_2$, while 20% were incorporated in biomass. In the GWSS slurry (10 g soil + 10 ml groundwater), NTA (initially 5 µg/l) was degraded with half-lives of 155 hours under anaerobic and 128 hours under aerobic conditions (Larson, 1984).

Shimp et al. (1994) assessed the mineralisation of $U^{14}C$-NTA in soil and groundwater samples from a septic tank system plume. The assays were incubated at 20°C, mineralisation of initially 25 – 100 ng NTA/l was measured as $^{14}CO_2$ evolution. In soil samples, activity was generally highest in the area immediately to the septic tank tile field, resulting in half-lives of < 3 days. At 20 m down gradient from the field, there was little biodegradative activity (not quantified). Similar results were obtained with groundwater samples: nearest the tile field a half-life of 0.78 days was determined, while in 40 m distance the half-life decreased to 9 days.

Degradation of NTA was investigated in a water/soil system simulation the underground passage (Stumpf et al., 1996). Samples with a soil/water ratio of 1/0.7 were dosed with 50 mg/l H_3NTA, incubated for 10 weeks, and the remaining NTA measured by GC. In the Ah- and Go horizons, after a lag-phase of 5 days the substance degraded rapidly ($t½$ = 1-2 days). In an aquifer sampled at an infiltration area of a drinking water work no lag phase was observed, indicating pre-adaption. The half-life was about 5 days, after 15 days the test substance was completely degraded. In all samples NTA metabolites were not detected.

The available studies demonstrate that NTA is mineralised in soils, half-lives between 1 – 9 days were determined. For the exposure calculations, a half-life of 9 days is used as a worst case.

3.1.2.1.2 Photolytical degradation

The Fe(III)NTA complex was found to undergo photolysis when exposed to sunlight. In a 1 mM aqueous solution irradiated with sunlight in July (t = 25 – 30°C), the concentration of NTA decreased rapidly over a period of 9 hours. As reaction products, iminodiacetate (IDA), CH_2O and CO_2 were detected. IDA is very slowly degraded to glycine under the same test conditions. Complexes with other metals were irradiated for a week: while no significant change was found for Pb(II) and Cd(II), a marginally decrease for Cr(III) and a marked (70%) decrease for Cu(II) was observed (Stolzberg and Hume, 1975).

Svenson et al. (1989) determined the quantum yield for Fe(III)NTA photolysis to 0.0129. For the yearly maximum of a solar spectrum at 60°N, a half-life of 42.9 minutes was calculated. This value refers to the top millimetres of a water body at noon, because of factors like cloudiness, shadowing effects of vegetation, absorption and scattering of light by suspended solids etc. the actual environmental lifetime is certainly longer.

The available studies demonstrate that in surface waters photolysis can contribute to NTA degradation. As referred in Section 3.1.3.5, in the thermodynamical equilibrium state there are only small fractions of the photolytically instable complex species. Biodegradation is the predominant degradation mechanism, thus photolysis is not considered in the exposure assessment.

Summary of degradation rates

The following degradation rates are used in the further exposure assessment:

Parameter	Degr. rate	Half-life
$kbio_{water}$	0.14 d^{-1}	5 days
$kbio_{sed}$ (aer)	0.077 d^{-1}	9 days
$kbio_{sed}$ (anaer)	0.035 d^{-1}	20 days
$kbio_{soil}$	0.077 d^{-1}	9 days
$kdeg_{air}$	0 d^{-1}	∞

3.1.2.2 Distribution

Because of the salt character of NTA no value for the vapour pressure is available; therefore Henry's law constant cannot be calculated from vapour pressure and water solubility. Volatilisation from aqueous solution is not expected. For the exposure calculations, the lowest value accepted by EUSES ($4 \cdot 10^{-10}$ Pa $\cdot$ m^3/mol) is used.

Due to the ionic structure under environmental relevant pH conditions, a relevant adsorption onto the organic fraction of soils or sediments is not expected. However, interaction with the mineral phase is possible.

The distribution of NTA between a marine surface sediment and a mineral medium (used for biodegradation tests) was examined by Bolton et al. (1993). After addition of 10 µM NTA, a mixture of complexes is formed. After 24 hours incubation at a temperature of 4°C, a distribution coefficient Kp of 1.6 l/kg was determined. In the exposure calculations, this value is used for adsorption onto both sediments, suspended particles, and soils.

3.1.2.3 Bioaccumulation

The uptake of NTA and its complexes by a series of species was examined in detail by Lentz and Lidzba (1988). With a Na_3NTA concentration of 400 µg/l, the following BCF values were obtained:

Table 3.3 BCF values for aquatic species

Species	BCF (l/kg)	Equilibrium after
Fish (*Brachidanio rerio*)	1 - 3	96 hours
Guppy (*Lebistes reticulatis*)	male 1 - 2 female 6	72 - 96 hours
Goldfish (*Carassius auratus*)	1 - 2	72 - 96 hours
Snail (*Lymnaea stagnalis*)	8 ≥ 20	3 – 7 days ≤ 72 days
Notonecta spec.	2 – 4	48 hours
Tubificidae	5 – 10	5 days
Frog larvae (*Rana temporaria*)	5 - 10	96 hours
Frog (*Rana temporaria*)	< 1	
Crayfish (*Procambarus*)	1	4 hours

In the same study, *Brachidanio rerio* was exposed to different NTA complexes. BCF values of 1.9, 3.4, and 2.8 were obtained for Cu-NTA, Cd-NTA, and FeNTA, respectively.

The available study demonstrates that only a low accumulation of NTA occurs in the hydrosphere. For the exposure calculations, a BCF value of 3 l/kg is chosen.

3.1.3 Aquatic compartment

3.1.3.1 Estimation of PEClocal during production

In the *Technical Guidance Documents for New and Existing Substances*, release factors into the raw sewage of 0.3% for production (TGD, Appendix I, Table A1.1) and 0.3% for formulation are proposed as default values.

In this section the exposure is calculated from specific data for 4 production sites: from the emission and production volumes, release factors from 0.4 ppm to 0.85% are calculated. In these factors waste water purification is included.

For calculating the $Clocal_{aqua}$, the dilution of the waste water in the river is considered according to

$$Clocal,water = Clocal,eff \cdot D \text{ with } D = Qww / Qriver$$

$C_{local,eff}$: concentration in WWTP effluent
D: dilution factor
Q_{ww}: sewage flow
Q_{river}: river flow (10%ile value preferred)

PECregional = 4.2 µg/l (see Section 3.1.3.6)

In **Table 3.4** the estimated concentrations, releases into the environment, and the underlying specific data (as far as available) are summarised:

Table 3.4 PEC calculation for Na_3NTA production

Site	Site-specific data	Defaults	Ceff (mg/l)	$Clocal_{aqua}$ (µg/l)	$PEClocal_{aqua}$ (µg/l)	Release
A	only import	-	0	0	-	0
B	Effluent conc.; release period 20 times/a for 24 hour, sewage flow	Dilution 1:10	No WWTP	< 0.93	< 5.1	<370 g/annum
C	Effluent conc.; sewage and river flow	-	0.054	1.4	5.6	7.6 tonnes/annum
D	Daily release, sewage flow		No WWTP	< 450	< 450	10.8 tonnes/annum
E	yearly release volume, production period; sewage and river flow	-	1.1	9.4	14	5.7 tonnes/annum

Site D The concentrations in the receiving estuary are modelled, resulting in maximum values in the range of 0.25 – 0.45 mg/l.
The total release at the production sites into surface waters is 24.1 tonnes/annum Na_3NTA.

3.1.3.2 Estimation of PEClocal during formulation and use

3.1.3.2.1 Textile Cleaning

NTA is an effective substitute for phosphates. It prevents calcium and magnesium ions from forming sparingly soluble salts with orthophosphate, pyrophosphate, carbonate, silicate and other inorganic anions as well as anionic surfactants and fatty acids (BASF, 2001).

In 2000, 973 tonnes of Na_3NTA were used for textile cleaning, in both household and industrial applications. As no breakdown between household and industrial use is available, as a worst case approach the exposure calculation is based on industrial use. Furthermore, it is assumed that releases occur only into waste water. All other parameters are taken from the Technical Guidance Documents. The 10% rule was applied due to the assumed wide and equal distribution of the sites over the EU (see **Table 2.1**).

Table 3.5 PEC calculation for textile cleaning

	Formulation	Use
Total use volume	973 tonnes/annum	
Continental use	876 tonnes/annum	
Regional use	97 tonnes/annum	
Fraction of local main source	1	0.4
Number of emission days	300 days/annum	194 days/annum
Fraction released to waste water	0.02	1
Release into untreated waste water	6.5 kg/day	200 kg/day
Release into hydrosphere (removal 95%)	0.32 kg/day	10 kg/day
Ceffluent	160 µg/l	5,000 µg/l
Clocal	16 µg/l	500 µg/l
PEClocal (PECregional = 4.2 µg/l)	20 µg/l	500 µg/l

3.1.3.2.2 Cleaning agents

NTA help prevent water-based formulations, especially neutral and alkaline liquid cleaners, from becoming cloudy or precipitating.

In 2000, 17,905 tonnes of Na_3NTA were used in cleaning agents for industrial applications. It is assumed that releases occur only into waste water. All other parameters are taken from the Technical Guidance Documents.

Table 3.6 PEC calculation for cleaning agents

	Formulation	Use
Total use volume	17,905 tonnes/annum	
Continental use	16,115 tonnes/annum	
Regional use	1,790 tonnes/annum	
Fraction of local main source	1	0.002
Number of emission days	300	200
Fraction released to waste water	0.003	1
Release into untreated waste water	17.9 kg/day	17.9 kg/day
Release into hydrosphere (removal 95%)	0.895 kg/day	0.895 kg/day
Ceffluent	450 µg/l	450 µg/l
Clocal	45 µg/l	45 µg/l
PEClocal (PECregional = 4.2 µg/l)	49 µg/l	49 µg/l

3.1.3.3 Sediments

No monitoring data for sediments are available. The Na_3NTA concentration could be modelled using the equilibrium partitioning method. As also no effect tests are available, a risk assessment for sediments would lead to identical PEC/PNEC ratios like for the aquatic compartment.

Because of the low partitioning coefficients, no accumulation in sediments is expected. Thus an assessment of this sub-compartment is not necessary.

3.1.3.4 Measured levels

In contrast to most publications about environmental fate of ecotoxicology where NTA amounts or concentrations are referred as sodium salt, monitoring data are generally referred as H_3NTA. In this section, the original data are cited. In order to receive figures comparable to the other sections of this report, the H_3NTA figures have to be multiplied with a factor of 1.35 to receive the Na_3NTA equivalents.

Monitoring in surface waters

NTA is continuously monitored in German surface waters (LAWA, 2000). In 1997/98, the substance was sampled at 84 locations at 51 rivers and creeks, mostly with 13 samples per year at each location. From a total of 2,283 measurements, the highest detected concentration was 100 µg/l H_3NTA (= 135 µg/l Na_3NTA). In the following table, the results are sorted in concentration ranges, the 90%ile values are considered:

Table 3.7 90%ile values of H_3NTA concentrations in German surface waters (LAWA, 2000)

90%ile concentration (µg/l)	Number of sites
0 - 1	2
1 – 10	65
10 - 100	17
Total	84

A survey of concentrations of NTA in UK rivers and at sewage treatment works was reported by FWR (1992). Sampling was carried out in April and May 1992 at 25 river sites and 10 sewage treatment works. NTA concentrations in river water ranged from less than the detection limit of 2 µg/l to 43 µg/l with a log normal distribution. In Class 3 rivers (NWC or Scottish Classifications, i.e., highly polluted rivers) the mean concentration was 16 µg/l and the median concentration was 10 µg/l. There were no identifiable differences between the means for the Class 1 or Class 2 rivers for which both the overall mean and the medians were less than 2 µg/l.

The following data were reported from an environmental survey in Austria: NTA was not detectable (detection limit: 2 µg/l) in 78 of 85 sampling places (number of samples for each location: 6). In six sample places, a maximum value between 6 and 10 µg/l was measured. In one case (sample from a channel near Vienna) the medium and highest values were 42 and 231 µg/l, respectively (FEA, 1996).

During a monitoring campaign in 10 Swiss rivers performed in 1990, the yearly averaged NTA concentrations were in the range of 0.8 – 10 µg/l (Giger et al., 1991). Measurements in 5 lakes near drinking water works performed in 1993 resulted in a maximum concentration of 0.5 µg/l (IAWR, 1993).

Monitoring in waste water

In the following table, an overview about measurements in waste water is presented. Generally there is no information about the origin of the NTA.

Table 3.8 NTA concentrations in in- and outflow streams from sewage plants

Location	Year	I/O	Conc. NTA (µg/l)	Reference
Zurich, Glatt	Winter 1984	Influent	40-380	Alder et al. (1990)
		Effluent	3-30	
	Winter 1987	Influent	330-1,490	
		Effluent	5-50	
Hessen	1987	Influent	100-300	Kröber and Häckl (1989)
		Effluent	<2-23	
Bielefeld-Heepen	1987	Influent	64-68	Lahl and Burbaum (1988)
		Effluent	8-16	
UK (10 plants)	1992	Effluent	< 2 – 740	FWR (1992)

Comparing the available monitoring data with the calculated PEC`s (see Section 3.1.3) it can be seen that there are mostly in the same order of magnitude. The calculated PEC for the use in

textile cleaning (see **Table 3.5**), which is based on a worst case approach, is clearly above the measured data.

3.1.3.5 NTA metal complexes in the hydrosphere

In natural waters, both natural and anthropogenic metals are present, which are able to form complexes with NTA. The environmental risk assessment is confronted with some problems (e.g. photolysis of complexes, remobilisation of sediment-bound metals, effects of heavy metals) which are related to the speciation under environmental conditions. Therefore the main features of the NTA complex chemistry have to be elaborated.

3.1.3.5.1 Stability of NTA complexes (Ringbom and Wänninen, 1979)

The most important property of NTA is to form complexes (usually 1:1-complexes) with multivalent metal ions. The stability of these complexes is usually described by the mass action law:

$$K_{MeZ} = \frac{[MeZ^{(m-n)-}]}{[Me^{n+}] \cdot [Z^{m-}]}$$

with $[MeZ^{(m-n)-}]$ the concentration of the metal complex

$[Me^{n+}]$ the concentration of the metal ion

$[Z^{m-}]$ the concentration of the NTA^{3-} anion (active complexing species)

K_{MeZ} the stability constant of the metal complex

The stability of the complexes of the $N(CH_2COO^{-})_3$ anion with a polyvalent metal ion is described by the stability constants listed in **Table 3.9** (K1 for 1 to 1 complexes and K2 for 2 to 1 complexes). As a result of polarisation of the OH bond in the chelate, the 1 to 1 complexes behave like weak acids and also dissociate. This effect is expressed by the dissociation constant Kd.

Table 3.9 Different stabilities of NTA chelates formed with various metal ions (Martell and Smith, 1974)

Metal ion	log K_1	log K_2	pK_d
Al^{3+}	11.4		5.09
Ca^{2+}	6.39	8.76	
Cd^{2+}	9.78	14.39	11.25
Co^{2+}	10.38	14.33	10.80
Cu^{2+}	12.94	17.42	9.14

Table 3.9 continued overleaf

Table 3.9 continued Different stabilities of NTA chelates formed with various metal ions (Martell and Smith, 1974)

Metal ion	log K1	log K2	pKd
Fe^{2+}	8.33	12.80	10.60
Fe^{3+}	15.90	24.30	4.1 (7.8[a])
Hg^{2+}	14.60		
Mg^{2+}	5.47		
Mn^{2+}	7.46	10.94	
Ni^{2+}	11.50	16.32	10.86
Pb^{2+}	11.34		
Zn^{2+}	10.66	14.24	10.06

a) Reacts as dibasic acid, ionic strength (25 0C) 0.1 M.

The distribution of the specific metal complexes in the hydrosphere cannot be derived directly from the mass action law, because of the following reasons:

- In aqueous solution, NTA can in principle occur as a neutral molecule or as ions with different charges. With increasing pH, ionisation increases and the formation of complexes is enhanced.
- Metals can form insoluble hydroxides (especially in alkaline medium), phosphates and carbonates, complexes with other ligands (e.g. humid substances) or can be adsorbed onto suspended solids, which decrease the concentration of free metal ions. Some of these reactions are also dependent on pH.
- Both effects are accounted by the conditional complex-formation constant. These constants pass for all metal complexes through a maximum as a function of pH value.

An example is iron that according to **Table 3.9** forms the most stable Fe(III)NTA complex ($\log K_1 = 15.9$). In addition iron is the most frequent transition metal in river water. This would suggest that the major product formed under environmental conditions is the Fe(III)NTA complex. Nevertheless, studies on the NTA speciation in surface waters (see below) reveal that no significant amounts of Fe(III)NTA are present in the thermodynamic equilibrium state, as insoluble $Fe(OH)_3$ and Fe(O)OH are formed which are adsorbed or form colloids.

Taking into account the different molecular weights of $N(CH_2COONa)_3$ (MW : 257 g/mol) and the metal ions, and on the assumption that a 1 to 1 complex is formed, 1 mg Na_3NTA hypothetically can bind the following amounts of metal ions in the optimum pH range (see **Table 3.10**):

Table 3.10 Complexing capacity of Na_3NTA

Metal	MW (g/mol)	Me. bound by 1 mg Na_3NTA (mg)
Transition metal		
Cu^{2+}	63.5	0.25
Zn^{2+}	65.4	0.25
Ni^{2+}	58.7	0.23
Fe^{3+}	55.8	0.22
Alkaline earth metal		
Ca^{2+}	40.1	0.16
Mg^{2+}	24.3	0.09

3.1.3.5.2 Exchange reactions of metal complexes

When metal complexes come into contact with other metals, metal exchange reactions occur. Complexes being wasted from any technical process reach a treatment plant and are mixed with other waste waters, leading to a change in the speciation. The same processes occur when the effluent is released into surface waters with a different metal composition. The mechanism of metal exchange reactions is dissociation of a metal complex with subsequent binding of another metal ion. The rate of the exchange reaction is limited by the rate of the dissociation of the mother complex. The reaction rates vary in a large range. For NTA, reaction rates in the order of hours for Ni and seconds for Zn are quoted (Bolton et al., 1996).

3.1.3.5.3 Effect on heavy metals in treatment plants

Raw sewages always contain a more or less high amount of heavy metals. In general heavy metals are strongly adsorbed onto sewage sludge thus being removed from the water phase. By complexation with agents like NTA the metals are kept in the soluble phase, when the chelator is biologically degraded the metal ions are set free and adsorbed. When the chelator is not completely degraded, in the effluent increased metal concentrations can occur.

The influence of NTA on heavy metal concentrations in a Swiss municipal treatment plant effluent was studied by Alder et al. (1990) by measurements of NTA and some heavy metals in the effluent. With a normal load (20 ± 15 µg/l H_3NTA in the effluent), remobilisation was found to be neglectible compared to the normal load. During a shock loading (up to 2,000 µg/l in the effluent), Zn and Pb were remobilised from sludge, and their effluent concentrations increased by 200 and 50%, respectively. The concentration of copper did not increase.

Similar results were obtained by Pöpel et al. (1984). The effluent concentrations of Ni, Pb, Zn were increased during shock loading (40 mg/l H_3NTA in the influent) when NTA degradation was disturbed. Under normal operation conditions the metal levels were not increased.

It can be concluded that increased heavy metal releases into the hydrosphere can occur with high NTA loads or when NTA degradation is disturbed. Simultaneously, the sludge deloaded is leading to a decreased contamination of agricultural soil when sludge is applied as fertiliser.

3.1.3.5.4 Speciation of NTA metal complexes in the hydrosphere

So far, no analytical methods are available to differentiate the individual metal complex species. During sample preparation, the metal complexes are destroyed, the sum of all species is determined, generally being expressed as amount of the uncomplexed agent.

A possibility to approximate the metal complex speciation is model calculations, which describe the state of thermodynamical equilibrium. With complex models, the chemical interactions and the competition between trace metals and different constituents in natural waters can be investigated. Naturally occurring chelators like humid acids can compete with the anthropogenic complexing agent, thus their inclusion into the calculation model is absolutely necessary. These models have to consider a large series of components, e.g.

- the major cations Ca, Mg, K, Na
- the trace metals Pb, Cu, Ni, Zn, Cd, Co, Hg, Mn, Fe
- the inorganic ligands CO_3, SO_4, Cl, F, Br, NH_3, PO_4, OH
- the adsorbent SiO_2 to represent suspended solids.

A speciation calculation using the model MULTI was conducted by Hennes and Eberle (1984), which studied the influence of H_3NTA (10 µg/l – 10 mg/l) on metal concentrations typically for the river Rhine. With 100 µg/l H_3NTA, Cu, Pb, and Ni are predominantly complexed, while above 500 µg/l Zn and Cd are strongly bound. With 1 mg/l NTA, the fraction of CaNTA increases to 85%. In the table below, the results for an NTA concentration of 100 µg/l are presented.

Different results were obtained by a speciation calculation predicted for the Swiss river Glatt. With a concentration of 2 µg/l, NTA is predominantly complexed with Ca, and only a small portion would be associated with heavy metals. The authors explain the discrepancy to the study of Hennes and Eberle (1984) by inclusion of natural ligands for Cu and Zn, which compete successfully with NTA for the metal ions (Bucheli-Witschel and Egli, 2001).

Table 3.11 Calculated Speciation of NTA in surface waters. Fractions [Me-NTA]/[NTA] in %

Metal	Hennes and Eberle (1984)	Bucheli-Witschel and Egli (2001)
Ca	25	95
Mg		3
Ni	25	
Zn	30	<1
Cu	15	

Attention has to be given to the fact that the modelled results are based on highly variable parameters like concentration of the complexing agent, metal composition of surface water etc. Furthermore, only the thermodynamical equilibrium state is modelled, degradation reactions are not considered. At the thermodynamic equilibrium, the complex formation is selective. The most preferred metals being complexed are those with the highest conditional complex-forming constants. With increasing concentrations, other metal ions are complexed successively. In German and Dutch rivers, heavy metal concentrations in the range of 10 - 20 µmol/l (predominantly Fe and Mn) are detected. The PECs for NTA are always lower, thus in the hydrosphere NTA is always completely complexed.

In seawater, complexation of heavy metals is not important because of the low NTA concentrations and the high Ca- and Mg levels (Bernhardt, 1991).

3.1.3.5.5 Influence on the partitioning of heavy metals in sediments and water

In the hydrosphere, presence of complexing agents can cause an increase in soluble levels of heavy metals. There is a tendency to remobilise the metals from highly loaded sediments which remain from high metal emissions in the past. On the other hand, adsorption of recently emitted heavy metals onto sediments and suspended solids could be prevented.

In a series of laboratory studies using sediment slurries remobilisation of adsorbed metals was observed. Dependent on the nature of sediments and the water phase, increasing metal concentrations were observed with NTA concentrations generally above 1 mg/l. The lowest concentrations were found with sediment from a sea harbour; remobilisation of Ni and Cu occurred with 0.1 mg/l and Zn with > 0.1 mg/l H_3NTA. In hard waters the remobilisation degree is substantially lower because of the competition of Ca (Bernhardt, 1991).

In real surface waters, the distribution of metals is not only determined by the complex formation properties, but also by a series of physical, chemical and microbiological interactions. The results of sediment slurry experiments describe the maximum remobilisation capacity of a complexing agent, i.e. a worst case situation.

An extended study on heavy metal remobilisation by NTA closer to natural conditions was performed by Lorenz (1997). Bottom-layered sediment bodies (8 natural lake and river sediments) were overflown by artificial river water containing 400 and 1,600 µg H_3NTA/l (i.e. 2.13 and 8.51 µmol/l), and the increase of heavy metals in the water phase was detected. During the test time (up to 42 days) NTA both aerobic and anaerobic degradation was observed with half-lives of about 6 and 20 days, respectively. Typical concentration patterns specific to each element were found, which generally were independent of the choice of sediment. At aerobic water phase (8 mg O_2/l) a 2 mm deep oxic capping sediment layer developed out of which an NTA depended remobilisation can occur. Under "worst case" conditions (highly polluted sediment from river Mulde, 2,000 µg/l Na_2HNTA) mainly Zn was temporary remobilised (1.5 µmol/l); while a significantly lower, temporary conversion for Pb (up to 0.3 µmol/l) was observed. The metal concentrations in water went through a well marked maximum after 3-5 days, i.e. the remobilisation process is reversible. No remobilisation of Fe, Mn, Cd, Cu, Ni, P or S was detected.

Anaerobic conditions were received by overflowing the sediment with oxygen-free water. No substantial remobilisation of the toxic metals Zn, Pb, Cd, Cu and Ni was observed. This is explained by formation of metal sulphides with an extreme low solubility. Thus, in nature even highly loaded sediments cannot release heavy metals as long as they are buried in deeper sediment layers. However, if anoxic sediment layers come into contact with oxygen (which occurs by whirling up during high water flows or by shipping traffic), sulphide is oxidised to sulfate and the metals are again available for remobilisation.

The author concludes that the extent of remobilisation is not primarily determined by the complex formation constants. Higher influences have the sediment load and the form of binding of the metals (Lorenz, 1997).

Within the framework of the NTA research program two approaches were selected in Germany. In the first experiment, the diffusion of heavy metals from an artificial sediment was determined in the presence and absence of free NTA in laboratory experiments (Donnert et al., 1991). From

resting (i.e. non suspended) kaolinite as the model sediment the addition of 850 µg/l of (free) H_3NTA duplicated the Zinc concentration in water. The process was developing very slowly and did not attain the equilibrium state after several hundred hours. A mathematical model based on the Langmuir sorption relationship was developed from this experiment for the description of the interaction between heavy metals, clay mineral layer and the water phase. The calculation was carried out for zinc according to the conditions of river Rhine, i.e., 250 hours contact time and the corresponding geometrical as well as chemical parameters like the initial zinc concentration of 230 µg/l in the water. The calculation predicts that approximately 10% (25 µg/l of Zn) of the initial zinc concentration can just be significantly remobilised by 200 µg/l of H_3NTA not yet bound on heavy metals.

In the second experiment, the concentrations of dissolved copper, nickel, and zinc were determined in a 1,000 m flume filled with river sediment over which water with or without the addition of uncomplexed NTA flowed at a slow rate (Bernhardt, 1991). A definite remobilisation of the widely distributed and easily remobilisable zinc began at a concentration of 50 µg/l of free NTA not yet bound on heavy metals. Other heavy metals, such as copper and nickel, were only remobilised at clearly higher NTA concentrations. In this flume experiment, an NTA end concentration of less than 20 µg/l of NTA was always obtained because of biodegradation on a flow section of approximately 900 m above a temperature of 10°C and independent of the initial NTA concentration (50 – 500 µg/l of NTA).

For the evaluation of the remobilisation process, the following topics need to be considered:

- NTA always occurs as metal complex in the hydrosphere. In German rivers, heavy metal concentrations in the order of range of 10 - 20 µmol/l (predominantly Fe and Mn) are detected. The stoichiometric Na_3NTA equivalent is 2.6 - 5.1 mg/l. All PECs are below this range. Therefore, all NTA is bound onto metals, and there is no free NTA available to remobilise metals from sediments. Only metal exchange reactions may occur.
- In remobilisation tests with freshly adsorbed heavy metals, some metals begin to remobilise when the NTA concentration exceeds 100 µg/l.
- Old highly loaded sediments remain in deeper (anoxic) sediment layers. Remobilisation from the deeper layers is limited by formation of nearly insoluble metal sulphides. Only if the sediments are whirled up during high water flows, a significant increase of heavy metal abundance in the water phase can occur.
- The more time has gone since the reduction of high metal emissions, the lower is the probability that these old sediments come into contact with the river water being available for metal remobilisation.
- It is not possible to give a single value for an NTA concentration at which no effects on metal remobilisation occurs. Because of the complexity of the NTA-metal interactions (dependent on metal concentrations, pH, nature of the sediment, concentration of organics etc.), it is not possible to come to a general rule for effects which is applicable to each river system. For individual surface waters, model calculations can be performed to receive a rough estimation.

It can be concluded that significant remobilisation processes can only occur in extreme cases, i.e. when high NTA amounts are released. This leads to an increase of metals with high conditional complex- formation constants. In this case they would be completely complexed with NTA. Simultaneously, the sediments are deloaded.

3.1.3.6 Regional exposure

For the regional exposure assessment it is assumed that the total European Na_3NTA consumption volume (26,642 tonnes in 2000) is released into the waste water. In accordance to the TGD, 10% of the European releases are taken for the regional and 90% for the continental scenario. In the following table, the release amounts and the resulting PECs of the EUSES calculation are presented (see http://ecb.jrc.it):

Table 3.12 Calculation of PECregional

Parameter	Regional	Contin.
Release into waste water [tonnes/annum]	2,664	23,978
Release into WWTPs (70%) [tonnes/annum]	1,865	16,785
Release via WWTP effluents [tonnes/annum]	93	839
Direct release into hydrosphere (30%) [tonnes/annum]	799	7,193
Total release into hydrosphere [tonnes/annum]	892	8,032
PECwater [µg/l]	4.2	0.48
PECair [µg/m^3]	$9.4 \cdot 10^{-16}$	$1.1 \cdot 10^{-16}$
PECagric.soil [µg/kg dw]	$6.9 \cdot 10^{-10}$	$7.9 \cdot 10^{-11}$
PECind.soil [µg/kg dw]	$2.6 \cdot 10^{-9}$	$3.0 \cdot 10^{-10}$
PECnat.soil [µg/kg dw]	$2.6 \cdot 10^{-9}$	$3.0 \cdot 10^{-10}$
PECsediment [µg/kg dw]	4.2	0.48

3.1.4 Atmosphere

No relevant releases into the atmosphere are expected.

3.1.5 Terrestrial compartment

No relevant releases into soils are expected.

3.1.6 Secondary poisoning

From the ionic structure of sodium NTA it can be concluded that a significant accumulation of these substances in the biota is not to be expected.

3.2 EFFECTS ASSESSMENT: HAZARD IDENTIFICATION AND DOSE (CONCENTRATION) - RESPONSE (EFFECT) ASSESSMENT

3.2.1 Aquatic compartment (incl. sediment)

A result of the exposure assessment was that in the environment over-stoichiometric amounts of metal ions are present, thus NTA is always complexed with metal ions. In Section 3.1.3.5, it is elaborated that always a mixture of different metal complex species occurs in surface waters. Similar reactions take place in toxicity test media, where metal ions are complexed when uncomplexed NTA is added. For the interpretation of the test results the complex speciation has to be considered.

In the effect tests, either H_3NTA or the sodium salt was used as test substance. In order to present comparable results, all effect values are referred as Na_3NTA.

3.2.1.1 Toxicity to fish

There is a large database on the toxicity of NTA on fish; an overview of the results considered to be valid is presented in **Table 3.13** and **3.14**.

NTA is a readily degradable substance, and in some test media NTA is possibly not stable throughout the test period. E.g. Macek and Sturm (1973) measured as an average 173 mg/l Na_3NTA in a medium with a nominal concentration of 200 mg/l, and 3.4 mg/l at nominal 5.6 mg/l. Therefore, only studies conducted with analytical monitoring are considered in this section.

Static and flow-through tests using the bluegill sunfish (*Lepomis macrochirus*) were conducted both in soft (60 mg/l $CaCO_3$) and hard water (170 mg/l $CaCO_3$). Monitoring showed essentially no loss of NTA throughout the test, which was not unexpected as the media were either sterile or were made from distilled water. The LC_{50} values in the static test system were 487 mg/l in hard water and 252 mg/l in soft water, while under flow-through conditions the values were 476 mg/l and 278 mg/l, respectively. Microscopic examinations of the fish gill tissues revealed slight damage consisting of the loss of lamellar interdigitation, the beginning of this effect was observed at 155 mg/l in the hard water static test, 115 mg/l in the soft water static test, 420 mg/l in the hard water flow-through and 370 mg/l in the soft water flow-through test (Weaver, 1970).

Table 3.13 Toxicity of Na_3NTA to fish in short-term tests

Species	Method	Test type	Test conditions			Exposure time	Effect conc. (mg/l)	Reference
			Temp (°C)	Hardness (mg/l $CaCO_3$)	pH			
Pimephales promelas	APHA method	Flow-through	25 ± 1	ND	7.9 – 9.2	96 hours	LC_{50} = 114	Arthur et al. (1974)
Pimephales promelas	APHA method	Flow-through	ND	35	ND	96 hours	LC_{50} = 127	Macek and Sturm (1973)
Lepomis macrochirus	APHA method	Static	20 ± 1	60 170	ND	96 hours	LC_{50} = 252 LC_{50} = 487	Weaver (1970)
Lepomis macrochirus	APHA method	Flow-through	20 ± 1	60 170	ND	96 hours	LC_{50} = 278 LC_{50} = 476	Weaver (1970)
Oncorhynchus mykiss	APHA method	Flow-through	ND	35	ND	96 hours	LC_{50} = 98	Macek and Sturm (1973)

All concentrations related to Na_3NTA and based on analytical measurements

Table 3.14 Toxicity of Na_3NTA to fish in long-term tests

Species	Method	Test type	Test conditions			Exposure time	Effect conc. (mg/l)	Reference
			Temp (°C)	Hardness (mg/l $CaCO_3$)	pH			
Lepomis macrochirus	APHA method	Flow-through	19 ± 0.5	35	7.1	28 days	LC_{10} = 96 LC_{55} = 173	Macek and Sturm (1973)
Pimephales promelas	APHA method	Flow-through	19 ± 0.5	35	7.1	28 days	LC_0 = 96 LC_{100} = 173	Macek and Sturm (1973)
Carassius auratus	Embryo-larval-test	Flow-through	18.2 – 25.8	50 200	7.9 – 8.1	8 days	LC_1 = 28.5 LC_{50} = 240 LC_1 = 30.1 LC_{50} = 243	Birge et al. (1979)
Ictalurus punctatus	Embryo-larval-test	Flow-through	25.9 – 29.6	50 200	7.9 – 8.1	9 days	LC_1 = <131 LC_{50} = 329 LC_1 = 138 LC_{50} = 385	Birge et al. (1979)
Oncorhynchus mykiss	Embryo-larval-test	Flow-through	12.5 – 14.5	50 200	7.9 – 8.1	27 days	LC_1 = <16.9 LC_{50} = 90.5 LC_1 = 20.2 LC_{50} = 114	Birge et al. (1979)
Pimephales promelas	Generation-cycle test	Flow-through	24 ± 2	34.0 – 45.2	7.6 ± 0.4	224 days	NOEC > 54	Arthur et al. (1974)

All concentrations related to $Na_3NTA$and based on analytical measurements

Macek and Sturm (1973) conducted the acute and long-term toxicity of NTA on different fish species. The test was performed in a flow-through system (detention time 5 hours) with a total hardness of 35 mg/l $CaCO_3$. 96-hour LC_{50} values of 98 mg/l for *Oncorhynchus mykiss* and 127 mg/l for *Pimephales promelas* were determined. After 28 days of exposure, for *Lepomis macrochirus* a LC_{10} of 96 mg/l and a LC_{55} of 173 mg/l was determined, while for *P. promelas* the LC_0 was 96 mg/l and the LC_{100} 173 mg/l. In the long-term test, both species cumulative mortality due to continuous NTA exposure was conspicuously absent. Examination of gills of fish exposed to 96 mg/l NTA for 28 days indicated no changes in histology.

Birge et al. (1979) conducted an embryo-larval-test with the fish species channel catfish (*Ictalurus punctatus*), goldfish (*Carassius auratus*), and rainbow trout (*Oncorhynchus mykiss,* formerly *Salmo gairdneri*). Each test was performed in a flow-through system (detention time 2.5 hours) at two water hardness levels (50 and 200 mg/l $CaCO_3$). The NTA concentration was monitored daily. Exposure was initiated 20 minutes after fertilisation in trout, 1 to 2 hours post-spawning for goldfish, and 2 to 12 hours after spawning for channel catfish. Average hatching times were 23, 4.5, and 4 days for trout, catfish, and goldfish. One test parameter was the egg hatchability, including all embryos (normal or aberrant). Another test parameter was the survival of normal organisms, determined at hatching and 4 days post-hatching. Normal organisms were defined as those animals that were free of gross teratic defects. At 4 days post-hatching, the Na_3NTA LC_{50} values were 90.5, 240.4, and 329.3 mg/l for trout, goldfish, and catfish stages exposed in soft water, and 114, 243.4, and 384.7 mg/l in hard water. The LC_1 values derived by log probit analysis were <16.9 mg/l (trout), 28.5 mg/l (goldfish), and <131 mg/l (catfish) in soft water, and 20.2, 30.1, and 138.4 mg/l in hard water.

An extended study of the acute and chronic toxicity of NTA on the fathead minnow (*Pimephales promelas)* according to the APHA standard procedure was conducted by Arthur et al. (1974). Water from Lake Superior (hardness about 40 mg/l as $CaCO_3$) was used as the test medium. Analytical measurements during the test period revealed that NTA was biodegraded (substance loss up to 24%), therefore all referred concentrations are based on the measurements. The short-term toxicity test was conducted in a flow-through test system (retention time 5 hours) with juvenile fish, the 96-hour LC_{50} was determined to 114 mg/l Na_3NTA. For the chronic (generation-cycle) test, twenty 3-15-day-old fry were placed in each vessel and exposed to 5 different NTA concentrations (2.1 – 53.9 mg/l Na_3NTA). After 30 days exposure, larval growth was not affected even by the highest tested concentration. After an exposure period of 224 days, there were no observable differences in survival, spawning activity, and egg hatchability at the highest tested concentration of 53.9 mg/l Na_3NTA. At this concentration, NTA is mainly complexed with Ca and Mg. The individual exposure time of single development stages is not definitively specified.

Tests on acute toxicity to fish resulted in 96-hour LC_{50} values in the range of 98 – 487 mg/l. In all of these tests effects were observed when the Na_3NTA concentration exceeded the stoichiometric metal levels (mainly Ca and Mg) in the medium. A generally accepted hypothesis is that the toxicological profile of complexing agents is based on disturbances of metal metabolism. It is expected that effects are caused by the uncomplexed agent. This is supported by the increased effect values in hard water. Even in the 28-day test with adult fish (Macek and Sturm, 1973) the LC_0 and LC_{10} values of 96 mg/l are approximately equal to the stoichiometric metal levels.

Lower effect values (LC_1 in the range of <16.9 – <131 mg/l) were determined by the embryo-larval tests conducted by Birge et al. (1979). The effect values are usually very low compared to effect values found by other authors. No explanation for these discrepancies could be found. A careful examination of the entire information provided by Birge et al. gave no

plausible reason for the inconsistency of the data. However, as it was not possible to reproduce the effect values, it was decided by the EU member states not to use these data for a derivation of a PNECaqua if other valid fish early life stage tests are available. Therefore, the effect values found by Birge et al. are not employed in the further effects assessment.

In a generation-cycle test over 224 days on *Pimephales promelas* (Arthur et al., 1974), there were no observable differences in survival, spawning activity, and egg hatchability at the highest tested concentration of 54 mg/l Na_3NTA (the active test substance was Ca- or Mg-NTA). Based in this study, the NOEC for fish is determined to 54 mg/l.

3.2.1.2 Toxicity to Invertebrates

Because of the ready degradability of NTA, in static tests the test solutions are probably not stable over the total test period. Based on the available monitoring data, it is expected that the stability is guaranteed up to 48 hours. Static or semi-static tests over a longer period are possibly not valid and thus not referred here. There is a large database on the toxicity of Na_3NTA on invertebrates; an overview of the results considered to be valid is presented in **Table 3.15** and **3.16**.

a) Crustacea

An immobilisation test on *Daphnia magna* in a medium with a water hardness of 286 mg/l $CaCO_3$ was conducted by Bringmann and Kühn (1977a). It is not clear whether H_3NTA or Na_3NTA was used as test substance. The test solution was neutralised (pH 7.6 – 7.7). After 24 hours exposure, an EC_0 of 800 mg/l, an EC_{50} of 950 mg/l, and an EC_{100} of 1,350 mg/l were obtained. In a further test (Bringmann and Kühn, 1982) in a similar medium without neutralisation, the effect concentrations were EC_0 75 mg/l, EC_{50} 79 mg/l, and EC_{100} 83 mg/l. After neutralisation (pH 8.0), the EC_{50} was above 1,000 mg/l. The test substance was not monitored, thus all concentrations are nominal. The results indicate that the effects were largely due to the change of the pH value.

A static short-term toxicity test with the crustacea *Daphnia magna* was carried out in a medium with a hardness of 220 mg/l $CaCO_3$ (Canton and Sloof, 1982). The EC_{50} value was in the range of 560 – 1,000 mg/l for Daphnia (endpoint: immobilisation, mortality).

Flannagan (1971) tested the toxicity of Na_3NTA on 17 species of macro-invertebrates using 4 different natural waters with different hardness. Monitoring of the test substance revealed no significant decrease over a period of 73 hours. *Hyallela azteci* was tested in a flow-through system in unbuffered water (pH 9.3, 21 mg/l $CaCO_3$), the 72-hour LC_{50} was above 250 mg/l. Experiments with *Gammarus lacustris* in a static system showed a LC_{50} of about 600 mg/l in unbuffered hard water. Tests with *Pontoporeia affinis* in a flow-through system, the LC_{50} was above 1,000 mg/l in buffered soft water (21 mg/l $CaCO_3$)

Table 3.15 Toxicity of Na_3NTA to invertebrates in short-term tests

Species	Method	Test type	Test conditions			Exposure time	Effect conc. (mg/l)	Nominal / measured	Reference
			Temp (°C)	Hardness (mg/l $CaCO_3$)	pH				
Crustacea									
Daphnia magna	Immobilisation test	Static	20 – 22	286	7.6 – 7.7	24 hours	EC_{50} = 950 *	N	Bringmann and Kühn (1977a)
Daphnia magna	Immobilisation test	Static	20	286	ND	24 hours	EC_{50} = 79 *	N	Bringmann and Kühn (1982)
Daphnia magna	Immobilisation test	Static	19 ± 1	220	ND	48 hours	EC_{50} = 560 – 1,000	N	Canton and Sloof (1982)
Hyallela azteci	APHA method	Flow-through	10	21	ND	72 hours	LC_{50} > 250	M	Flannagan (1971)
Pontoporeia affinis	APHA method	Flow-through	10	21	Ca. 7.0	96 hours	LC_{50} > 1,000	M	Flannagan (1971)
Gammarus lacustris	APHA method	Static	10	74	7.0	96 hours	LC_{50} = approx. 600	M	Flannagan (1971)
Gammarus pseudolimnaeus	APHA method	Flow-through	17 ± 1	ND	7.9 – 9.2	96 hours	LC_{50} = 98	M	Arthur et al. (1974)
Molluscs									
Physa heterostropha	APHA method	Static	20 ± 1	60 170	ND	96 hours	LC_{50} = 373 LC_{50} = 522	M	Weaver (1970)
Helisoma trivolis	APHA method	Static	Room t.	74	ND	96 hours	LC_{50} > 250	M	Flannagan (1971)
Physa spec.	APHA method	Static	Room t.	21 745	ND	96 hours	LC_{50} = approx. 400 LC_{50} = approx. 700	M	Flannagan (1971)
Insects									
Aedes aegypti	Dutch standard method / mortality	Static	23 ± 2	220	ND	48 hours	LC_{50} = > 5,600	N	Canton and Sloof (1982)

* Unclear whether concentrations refer to H_3NTA or Na_3NTA

Table 3.16 Toxicity of Na_3NTA to invertebrates in long-term tests

Species	Method	Test type	Test conditions			Exposure time	Effect conc. (mg/l)	Nominal / measured	Reference
			Temp (°C)	Hardness (mg/l $CaCO_3$)	pH				
Crustacea									
Gammarus pseudolimnaeus	Generation-cycle test	Flow-through	17 ± 1	35.2 – 45.4	7.8 ± 0.1	147 days	NOEC = 9.3	M	Arthur et al. (1974)
Snails									
Helisoma trivolis	Reproduction / mortality / growth / fecundity	Flow-through	23	ND	7.4 – 7.5	120 days	NOEC = 12.5	M	Flannagan (1974)

An extended study of the acute and chronic toxicity of NTA on the amphipod *Gammarus pseudolimnaeus* according to the APHA standard procedure was conducted by Arthur et al. (1974). Water from Lake Superior (hardness about 40 mg/l as $CaCO_3$) was used as the test medium. Analytical measurements during the test period revealed that NTA was biodegraded (substance loss up to about 50%); therefore all referred concentrations are based on the measurements. Both tests were conducted in a flow-through test system with a retention time of 5-6 hours. The 96-hour LC_{50} was determined to 98 mg/l Na_3NTA. For the chronic test, 25 individuals of 18-day-old newly hatched young were placed in each vessel and exposed to 5 different NTA concentrations (1.2 – 51.9 mg/l Na_3NTA). Over 21 weeks exposure, the recorded parameters were survival, final gravid females, number of produced young, total births, number of young per female, and births per female. No young or births were produced in the 18.7 and 51.9 mg/l test chambers. The reproduction index (determined by adding total births plus final gravid females divided by final number of surviving females) showed a significant decrease in female fecundity in concentrations ≥ 18.7 mg/l. The lowest tested concentration without significant effects was 9.3 mg/l Na_3NTA. At this concentration, NTA is mainly complexed with Ca and Mg.

b) Molluscs

Weaver (1970) conducted short-term tests with the snail *Physa heterostropha* in both soft and hard water. 10 adult snails with an average diameter of 1.0 cm were exposed for 96 hours. Monitoring showed essentially no loss of NTA throughout the test, which was not unexpected as the media were either sterile or were made from distilled water. The LC_{50} values were 522 mg/l in hard water (170 mg/l $CaCO_3$) and 373 mg/l in soft water (60 mg/l $CaCO_3$).

Flannagan (1971) tested the toxicity of Na_3NTA on 17 species of macro-invertebrates using 4 different natural waters with different hardness. Monitoring of the test substance revealed no significant decrease over a period of 73 hours. *Helisoma trivolis* was tested in both buffered and unbuffered media, in both experiments the LC_{50} was above 250 mg/l. At higher Na_3NTA concentrations, the toxicity was higher in the unbuffered medium; the lethal effects are probably due to the increase of pH. Experiments with *Physa sp*. resulted in LC_{50} values of about 400 mg/l in a water hardness of 21 mg/l $CaCO_3$ and about 700 mg/l in a water hardness of 745 mg/l $CaCO_3$.

A test on the influence of NTA on mortality, growth and fecundity through 4 generations of the freshwater snail *Helisoma trivolis* was conducted by Flannagan (1974). 10 juvenile snails were exposed to 5 Na_3NTA concentrations in a flow-through system, their weight was measured daily during the 120-day exposure period. 10 snails of the offspring of each concentration group were selected to form the next generation. Monitoring of NTA revealed that the test substance concentration was stable. No significant growth differences were found between snails exposed to 6.25 and 12.5 mg/l Na_3NTA and their controls. At 25 mg/l only the F1 snails were significantly smaller, while with 50 mg/l the F1 and F2 snails but not the F3 generation was smaller than the control groups. At 100 mg/l all four generations were reduced in weight. Lethal effects were not observed up to 100 mg/l. From this test, a NOEC of 12.5 mg/l can be determined.

c) Insects

A static short-term toxicity test with 3-4 weeks old larvae of the insect *Aedes aegypti* was carried out in a medium with a hardness of 220 mg/l $CaCO_3$ (Canton and Sloof, 1982). The LC_{50} was in the range of 5,600 – 10,000 mg/l.

All tests on acute toxicity to invertebrates showed effects only when the Na_3NTA concentration exceeded the stoichiometric metal levels of the medium. It is expected that effects are caused by the uncomplexed agent. This is supported by the increased effect values in hard water.

In long-term tests, the most sensitive organism was the amphipod *Gammarus pseudo limnaeus*. In a generation-cycle test over 21 weeks exposure, the lowest tested concentration without significant effects was 9.3 mg/l Na_3NTA. Based in this study, the NOEC for invertebrates is determined to 9.3 mg/l. At this concentration, NTA is mainly complexed with Ca and Mg.

3.2.1.3 Toxicity to algae

The influence of medium composition on the growth inhibition of 3 algal species (*Selenastrum capricornutum, Scenedesmus subspicatus, Chlorella vulgaris*) was examined by Millington et al. (1988). Bolds Basal medium (BBM) is a very rich medium containing much higher concentrations of nutrients compared to OECD and EPA media. The method used followed the OECD test guideline. NTA (unclear whether acid or sodium salt) was tested at 5, 10, 50, 80, and 100 mg/l. The 5-day NOECs (related to cell concentration) are 5 mg/l for all 3 species in both OECD and EPA medium, while 50 mg/l (*S. capricornutum*) and 80 mg/l (*S. subspicatus, C. vulgaris*) for BBM was obtained. The test results indicate that the apparent effects are mainly caused by nutrient deficiency.

Both static and continuous flow tests on growth inhibition of the diatom *Navicula seminulum* using hard and soft nutrient solution was conducted by Weaver (1970). Test cultures were prepared by placing diatom stock solution onto millipore filters and introducing the filters into flasks containing nutrient solutions. At the conclusion of each test the cultures were dried and weighed. Monitoring throughout the tests showed essentially no loss of NTA. In the static test, the 96-hour EC_{b50} were 477 mg/l for hard water and 185 mg/l for soft water. Similar results were obtained in the flow-through system, the 96-hour EC_{b50} were 477 mg/l for hard water and 133 mg/l for soft water. In both media, the concentrations of nutrient metals (e.g. 2 mg/l $ZnSO_4$ or 1 mg/l $CoCl_2$) were relatively high thus preventing nutrient deficiency.

A static growth inhibition test on *Chlorella vulgaris* and *Microcystis aeruginosa* was conducted by Canton and Slooff (1982). The 96-hour EC_{b50} for *C. vulgaris* is in the range of 560-1,000 mg/l and for *M. aeruginosa* in the range of 180 – 320 mg/l Na_3NTA. The concentrations of nutrient metals (e.g. 110 µg/l $ZnCl_2$ or 80 µg/l $CuSO_4$) in the test medium were relatively high thus preventing nutrient deficiency.

Table 3.17 Toxicity of Na_3NTA to algae in growth inhibition tests

Species	Medium/Test type	Test conditions			Exposure time	Effect conc. (mg/l)	Nominal / measured	Reference
		Temp (°C)	Hardness (mg/l $CaCO_3$)	pH				
Selenastrum capricornutum	BBM /static	22°C	ND	ND	5 days	NOEC = 50 mg/l	N	Millington et al. (1988)
	OECD /static					NOEC = 5 mg/l		
	EPA /static					NOEC = 5 mg/l		
Scenedesmus subspicatus	BBM /static	22°C	ND	ND	5 days	NOEC = 80 mg/l	N	Millington et al. (1988)
	OECD /static					NOEC = 5 mg/l		
	EPA /static					NOEC = 5 mg/l		
Chlorella vulgaris	BBM /static	22°C	ND	ND	5 days	NOEC = 80 mg/l	N	Millington et al. (1988)
	OECD /static					NOEC = 5 mg/l		
	EPA /static					NOEC = 5 mg/l		
Nacicula seminulum	APHA /static	20 ± 1	60 170	ND	96 hours	EC_{50} = 185 mg/l EC_{50} = 477 mg/l	M	Weaver (1970)
Nacicula seminulum	APHA/flow through	20 ± 1	60 170	ND	96 hours	EC_{50} = 133 mg/l EC_{50} = 477 mg/l	M	Weaver (1970)
Chlorella vulgaris	static	22 ± 2	220	ND	96 hours	EC_{50} > 560 mg/l	N	Canton and Sloof (1982)
Microcystos aeruginosa	static	23 ± 2	220	ND	96 hours	EC_{50} > 180 mg/l	N	Canton and Sloof (1982)
Cyclotella nana	static	20 ± 1	ND	8.2	72 hours	LOEC = 1 mg/l	N	Erickson et al. (1970)

The effect of NTA on the marine algae *Cyclotella nana* using synthetic seawater was studied by Erickson et al. (1970). Growth rates (determined as cell density) were determined with 0.25, 0.5, 1.0, 2.5, and 5.0 mg/l Na_3NTA after 72 hours. The nutrient metal concentrations are comparable with the OECD standard medium. Addition of 0.25 to 0.5 mg/l Na_3NTA resulted in greater growth at 8.5°C, but not at 2.0°C. The addition of 1.0 to 5.0 mg/l resulted in a progressive inhibitory effect with time (EC_{50} about 2.5 mg/l) at both temperatures. The inhibitory effect is attributed to the reduced bioavailibility of trace metals.

The apparent toxicity of complexing agents to algae in standard tests is related to essential trace metal bioavailibility. Trace metal levels tend to be more important in algal growth tests than in other short-term tests (e.g. on fish or daphnia); the main reason is the rapid increase of biomass during the test. In standard tests using uncomplexed agents, the concentrations of free essential metal ions decrease drastically, leading to nutrient deficiency and relatively low effect concentrations. Addition of higher amounts of nutrient metals result in detoxification of the agent. This effect is also known from other complexing agents, e.g. EDTA (Dufková, 1984) and [S,S]-Ethylenediamine disuccinate, [S,S]-EDDS (Schowanek et al., 1996), with both substances the apparent toxicity disappeared when stoichiometric amounts of the nutrient metals were added.

3.2.1.4 Effects on ecosystems – pond studies

The influence of 100 and 500 µg/l NTA (as H_3NTA) on the biocoenosis of pond ecosystems was investigated by Hamm (1991), and Kucklentz (1991). Eight experimental ponds were installed at a test field in Wielenbach, Germany. Each pond had a surface area of 1,700 – 2,000 m^2 and contained approximately 1,400 m^3 water. The ponds simulated a natural system with macrophytes and littoral vegetation, stocked with fresh water crayfish, 50 carps, 20 grass fish and 20 tenches (*Tinca tinca*). The winter population in all ponds was smaller than in summer. Two types of ponds were used, one having a minimal water exchange and a retention time of ca. six weeks, whereas the other type was equipped with a water-batcher, providing a continuous inflow of the test substance, with a retention time of exact two weeks. An average concentration of 100 µg/l NTA was measured in the stagnant pond while the two NTA flow ponds exhibited average inflow concentrations of 100 and 500 µg/l NTA. Phosphate (10 and 100 µg/l) and copper (50 µg/l), known as accelerating and inhibiting primary production served as positive controls. One pond of each type was not charged with test substances and served as negative control. One control and one test pond of each type is used. The experiments took place over two years (i.e., two vegetation periods). It was concluded that there was no significant difference in macrophytes, concentration of chlorophyll a, zoobenthos, zoo- or phytoplankton between the control ponds and the ponds with 100 or 500 g/l H_3NTA (i.e. 135 and 670 mg/l Na_3NTA) in summer or winter. No toxic effects were noted in the fish at both NTA levels. No remobilisation of metals took place.

3.2.1.5 Influence on the toxicity of heavy metals

The uptake of heavy metals by aquatic organisms is strongly dependent on the chemical speciation of the metal in the environment. Van Ginneken et al. (1999) demonstrated that NTA decreased the uptake of the radioisotopes ^{109}Cd and ^{65}Zn by the carp *Cyprinus carpio.* The presence of NTA (0.01 – 0.1 µmol/l) decreases the concentrations of free metal ion activity, thus the uptake was reduced.

Erickson et al. (1970) studied the influence of NTA on the toxicity of copper to the marine algae *Cyclotella nana*. In a growth inhibition test, extreme inhibition occurred after addition of 50 µg/l Cu, in two replicates growth was reduced to 11 and 17.5% of the control after 72 hours. Further addition of 0.25 to 5.0 mg/l H_3NTA reduced the growth inhibition, and a concentration of 0.5 mg/l was sufficient to nullify copper toxicity.

3.2.1.6 Toxicity to Microorganisms

A series of monospecies tests to microorganisms is available; an overview is presented in **Table 3.18**.

Table 3.18 Toxicity of Na_3NTA to microorganisms

Species	Method	Test type	Test conditions			Exposure time	Effect conc. (mg/l)	Nominal / measured	Reference
			Temp (°C)	Hardness (mg/l $CaCO_3$)	pH				
Bacteria/Cyanobacteria									
Pseudomonas fluorescens	Cell multiplication inhibition / biomass	Static	23 ± 2	220	ND	8 hours	EC_{50} = 3,200-5,600	N	Canton and Sloof (1982)
Pseudomonas putida	Cell multiplication inhibition / biomass	Static	27	ND	7.0	16 hours	EC_5 > 10,000	N	Bringmann and Kühn (1977b)
Protozoa									
Chilomonas paramaecium	Cell multiplication inhibition / biomass	Static	20	ND	6.9	72 hours	EC_5 > 540	N	Bringmann et al. (1980)
Entosiphon sulcatum	Cell multiplication inhibition / biomass	Static	20	ND	6.9	72 hours	EC_5 > 1,100	N	Bringmann (1978)
Uronema parduzci	Cell multiplication inhibition / biomass	Static	20	ND	6.9	72 hours	EC_5 > 1,100	N	Bringmann and Kühn (1980)

a) Bacteria

A static growth inhibition test on *Pseudomonas fluorescens* was conducted by Canton and Slooff (1982). The tested bacteria were in the log phase at the start of exposure. The medium contained 5,000 mg/l glucose as substrate. The 96-hour EC_{50} values are in the range of 3,200 – 5,600 mg/l.

Bringmann and Kühn (1977b) tested the inhibition of cell multiplication with *Pseudomonas putida*. No effects were observed in concentrations up to 10,000 mg/l H_3NTA (= 13,500 mg/l Na_3NTA) after 16 hours of exposure.

b) Protozoa

A test on cell multiplication inhibition with different protozoa was performed using identical experimental conditions. Stock and preliminary cultures of the test organisms were fed with living bacteria, whereas the test cultures were fed with inactivated bacteria. H_3NTA was used as test substance. After 72 hours exposure, the toxic threshold concentrations (EC_5) were > 400 mg/l H_3NTA (> 540 mg/l Na_3NTA) for *Chilomonas paramaecium* (Bringmann et al., 1980), > 800 mg/l H_3NTA (> 1,100 mg/l Na_3NTA) for *Entosiphon sulcatum* (Bringmann, 1978), and > 800 mg/l H_3NTA (> 1,100 mg/l Na_3NTA) for *Uronema parduzci* (Bringmann and Kühn, 1980).

3.2.2 Calculation of Predicted No Effect Concentration (PNEC)

3.2.2.1 Determination of PNECaqua

According to the results from different ecotoxicological studies discussed above, the toxicological profile of NTA is based on disturbances of metal metabolism. For the interpretation of toxicity tests, the complex formation properties of NTA have to be taken into account. In Section 3.1.3.5, the main features on complex chemistry in the environment are elaborated. The reactions in the test media are comparable.

Beside Ca and Mg, test media contain a certain amount of heavy metal ions being necessary as trace nutrients. The complex forming constants of heavy metal complexes are by several orders of magnitude higher than of Ca/Mg-complexes, thus after addition of the test substance NTA (as acid or Na-salt) the concentration of uncomplexed trace metals decreases drastically. The degree of Ca/Mg complexation is dependent on the amount of added NTA. Uncomplexed NTA is only present when it is present in over-stoichiometric concentrations.

The choice of the complex species being relevant for effect testing should consider the environmental relevance. Effect tests should be conducted with a complex for which metal toxicity can be excluded. As shown is Section 3.1.3.5, always a mixture of metal complexes is released or being formed in surface waters. Using the Ca-complex as test substance appears to be appropriate, as it is probably the predominant species in freshwater systems.

In tests on acute toxicity to fish effects were observed when the Na_3NTA concentration exceeded the stoichiometric metal levels (mainly Ca and Mg) in the medium. It is expected that effects are caused by the uncomplexed agent. In surface waters, always over-stoichiometric amounts of metal ions are present, thus the available tests are not relevant for environmental conditions. Even in the 28-day test with adult fish (Macek and Sturm, 1973) the LC_0 and LC_{10} values of 96 mg/l are approximately equal to the stoichiometric metal levels.

In a generation-cycle test on *P. promelas* over 224 days (Arthur et al., 1974), there were no observable differences in survival, spawning activity, and egg hatchability at the highest tested concentration of 54 mg/l Na_3NTA. At this concentration, NTA is mainly complexed with Ca and Mg. Based in this study, the NOEC for fish is determined to 54 mg/l.

Similar to the fish tests, all tests on acute toxicity to invertebrates showed effects only when the Na_3NTA concentration exceeded the stoichiometric metal levels of the medium. In hard water, the effect values are increased. It is expected that effects are caused by the uncomplexed agent, thus the available tests are not relevant for environmental conditions. In long-term tests, the most sensitive organism was the amphipod *Gammarus pseudolimnaeus*. In a generation-cycle test over 21 weeks exposure, the lowest tested concentration without significant effects was 9.3 mg/l Na_3NTA. At this concentration, NTA is mainly complexed with Ca and Mg. Based in this study, the NOEC for invertebrates is determined to 9.3 mg/l.

The apparent effects of complexing agents to algal growth are related to essential trace metal bioavailibility. Trace metal levels tend to be more important in algae tests than in short-term tests on fish or daphnia, the main reason is the rapid increase of biomass during the test. The effect concentrations increased with the trace metal amounts. The test results indicate that not the absolute NTA concentration, but rather the ratio of the NTA to the metal cation concentration is crucial to algae growth. In media with low trace metal concentrations like the OECD standard medium, effects were observed in the range of 1 – 5 mg/l, while in metal-enriched media the NOECs were ≥ 50 mg/l.

The apparent toxicity of complexing agents to algae can be caused either by its intrinsic toxicity or by indirect effects like nutrient deficiency. The studies cited above reveal that the effects are mainly caused by the latter. Therefore, this inhibition of algae growth is an artefact which is caused by the drastic increase of biomass during the test. Such indirect effects cannot be quantified from the laboratory tests, thus only theoretical considerations can be made. In German and Dutch rivers, heavy metal concentrations in the range of 10-20 µmol/l (predominantly Fe and Mn) are detected. The stoichiometric Na_3NTA equivalent, i.e. the NTA amount needed for complete complexation of the heavy metals, is 2.6 – 5.1 mg/l. Estimations of the speciation in the hydrosphere show that the largest NTA fraction is complexed with Ca, therefore complete complexation of heavy metals is expected only with extremely high NTA concentrations. As in the environment metal ions are generally present in over-stoichiometric amounts, nutrient deficiency is not expected. Nutrient deficiency in surface waters could only occur when essential metal ions are over-chelated. Furthermore, plant growth is influenced by many limiting parameters; probably the presence of macronutrients like phosphate or nitrate is of greater importance.

In addition to the discussed adverse effects like growth inhibition and nutrient deficiency, growth stimulating effects like eutrophication may occur. The presence of a chelator can improve the bioavailibility of nutrient metals. Also this effect can only qualitatively be assessed. In the environment, higher availability of trace elements through the complexing agent depends on the preloading of the water and could stimulate the processes of eutrophication. If trace elements like Fe, Co, Mn, and Zn are sufficiently available in a soluble form, the plants growth will not be influenced. Because of the presence of over-stoichiometric amounts of heavy metals, it is unlikely that eutrophication is caused by NTA.

Besides the monospecies tests used for the PNEC determination, a study on pond ecosystems (Kucklentz, 1991; Hamm, 1991) is available. With concentrations up to 500 µg/l H_3NTA (= 670 µg/l Na_3NTA) neither nutrient deficiency nor eutrophication was observed.

The effects assessment of NTA is based on long-term tests, which are available for fish, daphnids and algae. The most sensitive endpoint was found for the amphipod *Gammarus pseudolimnaeus* with a NOEC of 9.3 mg/l. According to TGD an assessment factor of 10 has to be used. Therefore, a PNECaqua of 0.93 mg/l is determined.

3.2.2.2 Determination of $PNEC_{micro-organism}$

There are tests on different microorganisms available. Similar to the algae tests, it cannot be excluded that the apparent effects in these tests were caused by nutrient deficiency by reduction of the concentrations of essential metals. It is expected that the intrinsic toxicity of NTA is lower. The available test results can be used as a worst case approach.

The lowest effect value obtained in a test with bacteria is > 3,200 mg/l for *Pseudomonas fluorescens*. For the PNEC derivation this value is not employed as glucose was used as substrate. The lowest effect value from protozoa tests is a 72-hour EC_5 of > 540 mg/l (as Na_3NTA) for *Chilomonas paramaecium*. In accordance to the Technical Guidance Documents, an assessment factor of 1 is applied.

→ PNECmicroorg. > 540 mg/l

3.2.3 Atmosphere

Because there are no fumigation tests available, an effects assessment for this compartment can not be performed.

3.2.4 Terrestrial compartment

There are no tests on terrestrial organisms available, thus an effects assessment for this compartment can not be performed.

3.2.5 Secondary poisoning

As there is no bioaccumulation, a biomagnification via the food chain is not expected.

3.3 RISK CHARACTERISATION

3.3.1 Aquatic compartment

The risk assessment for aquatic organisms resulted in a PNECaqua of 0.93 mg/l. The PNECmicroorg. was determined to > 540 mg/l.

In the following table, the results for all calculated exposure scenarios are listed:

Table 3.19 Risk characterisation for the aquatic compartment

Scenario	$PEClocal_{aqua}$ (µg/l)	PECaqua / PNECaqua	Ceffl (mg/l)	Ceffl. / PNECmicro
Producer A	Only import			
Producer B	< 5.1	< 0.005	No WWTP	
Producer C	5.6	0.006	0.054	< 0.0001
Producer D	< 450	< 0.48	No WWTP	
Producer E	14	0.015	1.1	< 0.002
Textile Cleaning, Formulation	20	0.022	0.16	< 0.0003
Textile Cleaning, Use	500	0.54	5.0	< 0.009
Cleaning agents, Formulation	49	0.053	0.45	< 0.0008
Cleaning agents, Use	49	0.053	0.45	< 0.0008

The available studies on biodegradation reveal that NTA is removed in municipal treatment plants with rates generally above 95% under normal operation conditions. Contradicting results were obtained for measurements in the winter season: while in several cases no difference between summer and winter was observed, other studies show considerable decrease (< 50%) at low temperatures. For the exposure calculations in this assessment, a removal of 95% was used. Assuming in a worst case approach a removal rate of 50%, only the PEC/PNEC ratio of the scenario for the use in textile cleaning will be > 1 (2.0). Taking into account, that this scenario is nearly completely based on default values, it is concluded that even with a lower biodegradation there is no risk to the aquatic compartment.

Sediments

There are neither monitoring data for sediments nor toxicity tests with benthic organisms available. The Na_3NTA concentration could be modelled using the equilibrium partitioning method. A risk assessment for sediments would lead to identical PEC/PNEC ratios like for the aquatic compartment.

Because of the low partitioning coefficients, no accumulation in sediments is expected. Thus an assessment of this sub-compartment is not necessary.

Influence on the Distribution of Heavy Metals

In Section 3.1.3.5 the influence of NTA on the distribution of heavy metals was examined. It was concluded that significant remobilisation processes are only expected in extreme cases, i.e. when high NTA amounts are released. This would lead to increased concentrations of those metals with high conditional complex-formation constants. With the concentrations estimated in this risk assessment, those effects are not expected.

Conclusion (ii)

3.3.2 Atmosphere

No relevant releases into the atmosphere are expected; therefore a risk characterisation for this compartment is not necessary.

Conclusion (ii)

3.3.3 Terrestrial compartment

No relevant releases into soils are expected; therefore a risk characterisation for the terrestrial compartment is not necessary.

Conclusion (ii)

3.3.4 Secondary Poisoning

As there is no bioaccumulation, a biomagnification via the food chain is not expected.

Conclusion (ii)

4 HUMAN HEALTH

(to be added later)

5 RESULTS

5.1 ENVIRONMENT

Conclusion (ii) There is at present no need for further information and/or testing and for risk reduction measures beyond those which are being applied already.

In the present risk assessment production and use of Na_3NTA are examined. For all life-cycle steps, the PEC/PNEC ratios are below 1. Therefore, a risk for the environment is not expected.

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ABBREVIATIONS

ADI	Acceptable Daily Intake
AF	Assessment Factor
ASTM	American Society for Testing and Materials
ATP	Adaptation to Technical Progress
AUC	Area Under The Curve
B	Bioaccumulation
BBA	Biologische Bundesanstalt für Land- und Forstwirtschaft
BCF	Bioconcentration Factor
BMC	Benchmark Concentration
BMD	Benchmark Dose
BMF	Biomagnification Factor
BOD	Biochemical Oxygen Demand
bw	body weight / *Bw*, *bw*
C	Corrosive (Symbols and indications of danger for dangerous substances and preparations according to Annex II of Directive 67/548/EEC)
CA	Chromosome Aberration
CA	Competent Authority
CAS	Chemical Abstract Services
CEC	Commission of the European Communities
CEN	European Standards Organisation / European Committee for Normalisation
CEPE	European Committee for Paints and Inks
CMR	Carcinogenic, Mutagenic and toxic to Reproduction
CNS	Central Nervous System
COD	Chemical Oxygen Demand
CSTEE	Scientific Committee for Toxicity, Ecotoxicity and the Environment (DG SANCO)
CT_{50}	Clearance Time, elimination or depuration expressed as half-life
d.wt	dry weight / dw
dfi	daily food intake
DG	Directorate General
DIN	Deutsche Industrie Norm (German norm)
DNA	DeoxyriboNucleic Acid
DOC	Dissolved Organic Carbon
DT50	Degradation half-life or period required for 50 percent dissipation / degradation
DT90	Period required for 90 percent dissipation / degradation
E	Explosive (Symbols and indications of danger for dangerous substances and preparations according to Annex II of Directive 67/548/EEC)
EASE	Estimation and Assessment of Substance Exposure Physico-chemical properties [Model]

EbC50	Effect Concentration measured as 50% reduction in biomass growth in algae tests
EC	European Communities
EC10	Effect Concentration measured as 10% effect
EC50	median Effect Concentration
ECB	European Chemicals Bureau
ECETOC	European Centre for Ecotoxicology and Toxicology of Chemicals
ECVAM	European Centre for the Validation of Alternative Methods
EDC	Endocrine Disrupting Chemical
EEC	European Economic Communities
EINECS	European Inventory of Existing Commercial Chemical Substances
ELINCS	European List of New Chemical Substances
EN	European Norm
EPA	Environmental Protection Agency (USA)
ErC50	Effect Concentration measured as 50% reduction in growth rate in algae tests
ESD	Emission Scenario Document
EU	European Union
EUSES	European Union System for the Evaluation of Substances [software tool in support of the Technical Guidance Document on risk assessment]
F(+)	(Highly) flammable (Symbols and indications of danger for dangerous substances and preparations according to Annex II of Directive 67/548/EEC)
FAO	Food and Agriculture Organisation of the United Nations
FELS	Fish Early Life Stage
foc	Organic carbon factor (compartment depending)
GLP	Good Laboratory Practice
HEDSET	EC/OECD Harmonised Electronic Data Set (for data collection of existing substances)
HELCOM	Helsinki Commission -Baltic Marine Environment Protection Commission
HPLC	High Pressure Liquid Chromatography
HPVC	High Production Volume Chemical (> 1000 tonnes/annum)
IARC	International Agency for Research on Cancer
IC	Industrial Category
IC50	median Immobilisation Concentration or median Inhibitory Concentration
ILO	International Labour Organisation
IPCS	International Programme on Chemical Safety
ISO	International Organisation for Standardisation
IUCLID	International Uniform Chemical Information Database (existing substances)
IUPAC	International Union for Pure and Applied Chemistry
JEFCA	Joint FAO/WHO Expert Committee on Food Additives
JMPR	Joint FAO/WHO Meeting on Pesticide Residues

Koc	organic carbon normalised distribution coefficient
Kow	octanol/water partition coefficient
Kp	solids-water partition coefficient
L(E)C50	median Lethal (Effect) Concentration
LAEL	Lowest Adverse Effect Level
LC50	median Lethal Concentration
LD50	median Lethal Dose
LEV	Local Exhaust Ventilation
LLNA	Local Lymph Node Assay
LOAEL	Lowest Observed Adverse Effect Level
LOEC	Lowest Observed Effect Concentration
LOED	Lowest Observed Effect Dose
LOEL	Lowest Observed Effect Level
MAC	Maximum Allowable Concentration
MATC	Maximum Acceptable Toxic Concentration
MC	Main Category
MITI	Ministry of International Trade and Industry, Japan
MOE	Margin of Exposure
MOS	Margin of Safety
MW	Molecular Weight
N	Dangerous for the environment (Symbols and indications of danger for dangerous substances and preparations according to Annex II of Directive 67/548/EEC
NAEL	No Adverse Effect Level
NOAEL	No Observed Adverse Effect Level
NOEL	No Observed Effect Level
NOEC	No Observed Effect Concentration
NTP	National Toxicology Program (USA)
O	Oxidising (Symbols and indications of danger for dangerous substances and preparations according to Annex II of Directive 67/548/EEC)
OC	Organic Carbon content
OECD	Organisation for Economic Cooperation and Development
OEL	Occupational Exposure Limit
OJ	Official Journal
OSPAR	Oslo and Paris Convention for the protection of the marine environment of the Northeast Atlantic
P	Persistent
PBT	Persistent, Bioaccumulative and Toxic
PBPK	Physiologically Based PharmacoKinetic modelling

PBTK	Physiologically Based ToxicoKinetic modelling
PEC	Predicted Environmental Concentration
pH	logarithm (to the base 10) (of the hydrogen ion concentration $\{H^+\}$
pKa	logarithm (to the base 10) of the acid dissociation constant
pKb	logarithm (to the base 10) of the base dissociation constant
PNEC	Predicted No Effect Concentration
POP	Persistent Organic Pollutant
PPE	Personal Protective Equipment
QSAR	(Quantitative) Structure-Activity Relationship
R phrases	Risk phrases according to Annex III of Directive 67/548/EEC
RAR	Risk Assessment Report
RC	Risk Characterisation
RfC	Reference Concentration
RfD	Reference Dose
RNA	RiboNucleic Acid
RPE	Respiratory Protective Equipment
RWC	Reasonable Worst-Case
S phrases	Safety phrases according to Annex IV of Directive 67/548/EEC
SAR	Structure-Activity Relationships
SBR	Standardised birth ratio
SCE	Sister Chromatic Exchange
SCHER	Scientific Committee on Health and Environmental Risks
SDS	Safety Data Sheet
SETAC	Society of Environmental Toxicology And Chemistry
SNIF	Summary Notification Interchange Format (new substances)
SSD	Species Sensitivity Distribution
STP	Sewage Treatment Plant
T(+)	(Very) Toxic (Symbols and indications of danger for dangerous substances and preparations according to Annex II of Directive 67/548/EEC)
TDI	Tolerable Daily Intake
TG	Test Guideline
TGD	Technical Guidance Document
TNsG	Technical Notes for Guidance (for Biocides)
TNO	The Netherlands Organisation for Applied Scientific Research
ThOD	Theoritical Oxygen Demand
UC	Use Category
UDS	Unscheduled DNA Synthesis
UN	United Nations

UNEP	United Nations Environment Programme
US EPA	Environmental Protection Agency, USA
UV	Ultraviolet Region of Spectrum
UVCB	Unknown or Variable composition, Complex reaction products of Biological material
vB	very Bioaccumulative
VOC	Volatile Organic Compound
vP	very Persistent
vPvB	very Persistent and very Bioaccumulative
v/v	volume per volume ratio
w/w	weight per weight ratio
WHO	World Health Organisation
WWTP	Waste Water Treatment Plant
Xn	Harmful (Symbols and indications of danger for dangerous substances and preparations according to Annex II of Directive 67/548/EEC)
Xi	Irritant (Symbols and indications of danger for dangerous substances and preparations according to Annex II of Directive 67/548/EEC)

European Commission

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Trisodium nitrilotriacetate - Part I - Environment, Volume 60

Editors: S.J. Munn, R. Allanou, K. Aschberger, O. Cosgrove, S. Pakalin, A. Paya-Perez, G. Pellegrini, B. Schwarz-Schulz, S. Vegro

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Environment and quality of life series

The report provides the comprehensive risk assessment of the substance trisodium nitrilotriacetate. It has been prepared by Germany in the frame of Council Regulation (EEC) No. 793/93 on the evaluation and control of the risks of existing substances, following the principles for assessment of the risks to humans and the environment, laid down in Commission Regulation (EC) No. 1488/94.

Part I – Environment

This part of the evaluation considers the emissions and the resulting exposure to the environment in all life cycle steps. Following the exposure assessment, the environmental risk characterisation for each protection goal in the aquatic, terrestrial and atmospheric compartment has been determined.

The environmental risk assessment for trisodium nitrilotriacetate concludes that there is at present no concern for the environment.

Part II – Human Health

This part of the report is not yet completed and will be published later.

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European Union Risk Assessment Report

trisodium nitrilotriacetate
Part I - environment

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