oekotoxzentrum
centre ecotox

Schweizerisches Zentrum für angewandte Ökotoxikologie
Centre Suisse d'écotoxicologie appliquée

# SQC (EQS~sed~) – Proposal from the Ecotox Centre for: *6-Acetyl-1,1,2,4,4,7-hexamethyltetraline (AHTN), "Tonalid"*

## Imprint

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**Authors**

Alexandra Kroll, Carmen Casado-Martinez        Swiss Centre for Applied Ecotoxicology

**Scientific Support**

Dr Karen Duis, ECT Oekotoxikologie GmbH, Böttgerstr. 2-14, D-65439 Flörsheim/Main, Germany

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**Contact**

Alexandra Kroll: Alexandra.Kroll@oekotoxzentrum.ch
Carmen Casado-Martinez: Carmen.Casado@centreecotox.ch

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## Summary

**SQC (EQS$_{sed}$):**        **344 µg/kg d.w.**

In the framework of the Module Sediment, which is intended to help cantons in sediment quality assessment, the Ecotox Centre develops proposals for Environmental Quality Criteria for sediment (SQC). SQC are derived applying the methodology described in the EU-Technical Guidance (TGD) for Deriving Environmental Quality Standards (EQS). In order to ensure that the dossiers are internationally comparable, the English terminology of the TGD will be used in the remainder of the dossier. These criteria provide a first screening tool to evaluate sediment chemical quality and the potential risk for the aquatic ecosystem. Based on the scientific literature available at present a generic SQC for 6-Acetyl-1,1,2,4,4,7-hexamethyltetraline (AHTN), "Tonalid" of 344 µg/kg d.w. is proposed for standard sediments with 1 % OC.

## Zusammenfassung

**SQK (EQS$_{sed}$):**        **344 µg/kg TS**

Im Rahmen des Sedimentmoduls, das den Kantonen bei der Bewertung der Sedimentqualität helfen soll, entwickelt das Oekotoxzentrum Vorschläge für Umweltqualitätskriterien für Sedimente (SQK). Diese Kriterien dienen als Methode für ein erstes Screening zur Bewertung der chemischen Sedimentqualität und des potenziellen Risikos für aquatische Ökosysteme. Auf der Basis von Literaturdaten für die Wirkung von 6-Acetyl-1,1,2,4,4,7-hexamethyltetraline (AHTN), "Tonalid" und unter Verwendung der Methode, die in der Technischen Richtlinie der EU zur Ableitung von Umweltqualitätsnormen beschriebenen wird, schlägt das Oekotoxzentrum einen allgemeines SQK für 6-Acetyl-1,1,2,4,4,7-hexamethyltetraline (AHTN), "Tonalid" von 344 µg/kg TS für Standardsedimente mit 1 % OC vor.

## Résumé

**CQS (EQS$_{sed}$):**        **344 µg/kg p.s.**

Dans le cadre du module Sédiments qui devrait aider les cantons à évaluer la qualité des sédiments, le Centre Ecotox élabore des propositions de critères de qualité environnementale pour les sédiments (CQS). Les CQS sont dérivés en appliquant la méthodologie décrite dans le Guide Technique de l'UE (TGD) pour la Dérivation des Normes de Qualité Environnementale (EQS). Afin que les dossiers soient comparables au niveau international, la terminologie anglaise du TGD est utilisée ci-dessous. Ces critères fournissent un premier outil de dépistage pour évaluer la qualité chimique des sédiments et le risque potentiel pour l'écosystème aquatique. Sur la base des données sur les effets existants dans la littérature un CQS générique pour le 6-Acetyl-1,1,2,4,4,7-hexamethyltetraline (AHTN), "Tonalid" de 344 µg/kg p.s. est proposé pour les sédiments standards avec 1 % CO.

## Sommario

**CQS (EQS$_{sed}$):**     **344 µg/kg p.s.**

Nell'ambito del modulo Sedimenti, che è finalizzato ad aiutare i Cantoni nella valutazione della qualità dei sedimenti, il Centro Ecotox sviluppa proposte per i criteri di qualità ambientale per i sedimenti (CQS). I CQS sono derivati applicando la metodologia descritta nella Guida Tecnica dell'UE (TGD) per la Derivazione degli Standard di Qualità Ambientale (EQS). Per garantire che i dossier siano comparabili a livello internazionale, viene utilizzata la terminologia inglese del TGD. Questi criteri forniscono un primo strumento di screening per valutare la qualità chimica dei sedimenti e il potenziale rischio per l'ecosistema acquatico. Sulla base della letteratura scientifica disponibile allo stato attuale un CQS generico per il 6-Acetyl-1,1,2,4,4,7-hexamethyltetraline (AHTN), "Tonalid" di 344 µg/kg p.s. è proposto per sedimenti standard con 1 % CO.

# Table of content

# 1   General Information

Selected information on the polycyclic synthetic musk 6-Acetyl-1,1,2,4,4,7-hexamethyltetraline (AHTN) relevant for sediment is presented in this chapter. Registration information and risk assessments referred to are:

- Tonalide and related polycyclic musks: Environment tier II assessment (Australian Government 2016)
- Environmental risk assessment for the polycyclic musks AHTN and HHCB in the EU: I. Fate and exposure assessment (Balk & Ford 1999a)
- Environmental risk assessment for the polycyclic musks, AHTN and HHCB. II. Effect assessment and risk characterisation (Balk & Ford 1999b)
- European Union Risk Assessment Report 1-(5,6,7,8-TETRAHYDRO-3,5,5,6,8,8-HEXAMETHYL-2-NAPTHYL)ETHAN-1-ONE (AHTN), CAS No: 1506-02-1 or 21145-77-7, EINECS No: 216-133-4 or 244-240-6 (EC 2008)
- Information on Registered Substances (ECHA 2012a), (ECHA 2012b); EU risk assessment only performed for the second entry
- Polycyclic musks AHTN (CAS 1506-02-1) and HHCB (CAS 1222-05-05) Environmental Section, Version 2.0 (HERA 2004)
- OSPAR background document on musk xylene and other musks. ISBN 1-904426-36-0. (OSPAR Commission 2004)

## 1.1   Identity and physico-chemical properties

Two different CAS and EINECS numbers exist for AHTN which is due to an error in the molecular structure for AHTN reported by one company and a correct molecular structure by a second company. The molecular structure has been corrected (EC 2008). Trade names of AHTN are Fixolid, Ganolid, Kevolide, Tentarome, Tetralide, and Tonalid/Tonalide with Tonalid(e) being the trade name most often used in the literature.
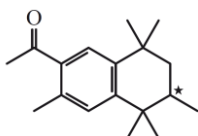
The molecular structure of AHTN has one stereogenic centre resulting in two enantiomers with a ratio of 1:1 in technical AHTN (EC 2008).

The log $K_{oc}$ reported for AHTN are in the range of 3.76-4.9 (Table 1, Appendix I). Reported experimental log $K_{ow}$ range from 5.4-5.7 (Table 1; geometric mean = 5.55), estimated log $K_{ow}$ range from 5.32-6.35 (Table 1). Both parameters trigger an effects assessment for sediments according to the EC TGD EQS (EC 2018).

Table 1 summarizes identity and physico-chemical parameters for AHTN required for EQS derivation according to the TGD (EC 2018). Where available, experimentally collected data is identified as (exp.) and estimated data as (est.). When not identified, no indication is available in the cited literature.

*Table 1 Information required for EQS derivation according to the TGD (EC 2018).*

| Characteristics | Values | References |
|---|---|---|
| Common name | 1-(5,6,7,8-Tetrahydro-3,5,5,6,8,8-hexamethyl-2-napthyl)ethan-1-one<br>2'–Acetonaphtone,5',6',7',8'-tetrahydro-3',5',5',6',8',8'-hexamethyl<br>6-Acetyl-1,1,2,4,4,7-hexamethyl-1,2,3,4-tetrahydronaphtalene<br>**6-Acetyl-1,1,2,4,4,7-hexamethyltetraline**<br>7-Aceto-1,1,3,4,4,6-hexamethyltetraline<br>7-Aceto-1,2,3,4-tetrahydro-1,1,3,4,4,6-hexamethylnaphtalene<br>7-Acetyl-1,1,3,4,4,6-hexamethyl-1,2,3,4-tetrahydronaphtalene<br>AHMT<br>**AHTN**<br>Ethanone,1-(5,6,7,8-tetrahydro-3,5,5,6,8,8-hexamethyl-2-naphtalenyl)-<br>Fixolide (trade name)<br>Tentarome (trade name)<br>Tetralide (trade name)<br>Tonalid (trade name) | EC (2008) |
| IUPAC name | 6-Acetyl-1,1,2,4,4,7-hexamethyltetraline | EC (2008) |
| Chemical group | Hydronaphthalenes | |
| Structural formula | <br>* stereogenic centre | Legrum (2011) |
| Molecular formula | C18H26O | EC (2008) |
| CAS | 1506-02-1 or 21145-77-7 | EC (2008) |
| EC Number | 216-204-6 or 244-240-6 | EC (2008) |
| SMILES code | CC1CC(C2=C(C1(C)C)C=C(C(=C2)C(=O)C)C)(C)C | PubChem (2019) |
| Molecular weight [g/mol] | 258.4 | Balk & Ford (1999a) |
| Melting point [°C] | [1] > 54 °C (exp., method unknown)<br>[2] 54 - 56°C (97.0-99.2% purity; exp., capillary method) | [1] Janssen, 2004a cited in EC (2008)<br>[2] ECHA (2012b) |
| Boiling point [°C] | 326 °C at 1 atm (exp., method unknown) | Janssen, 2004b cited in EC (2008) |
| Vapour pressure [Pa] | 0.0682 Pa at 25 °C (exp., gas saturation method, OECD TG 104, ^{14}C-labelled material) | MacGillivray, 1996 cited in EC (2008) |
| Henry's law constant [Pa·m$^3$/mol] | [1] 12.5 (exp.)<br>[2] 37.1 at 25 °C (exp.) | [1] Balk & Ford (1999a) |

| | | |
|---|---|---|
| | | [2] Artola-Garciana, 2002 cited in EC (2008) |
| Water solubility  [mg/L] | [1] 1.25 at 25 °C (exp., flask method, OECD TG 105, $^{14}$C-labelled material) <br> [2] 0.91 (± 0.04) (exp., column elution method) | [1] Edwards, 1996, cited in EC (2008) <br> [2] Artola-Garciana, 2002, cited in EC (2008) |
| Dissociation constant (pK$_a$) | Not relevant[b] | |
| Octanol-water partition coefficient (log K$_{ow}$) [a] | [1] 5.7 (Fixolide; exp., reversed-phase HPLC, OECD TG 117) <br> [2] 5.7 (est., EPIsuite v. 4.0) <br> [3] 5.4 (exp., slow stirring method) <br> [4] 6.35 (est., SRC, version 1.57) <br> [5] 6.25 (est., Biobyte ClogP 4.01) <br> [6] 5.32 (est., Molinspiration property engine v2018.10) | [1] Rudio, 1993a cited in EC (2008) <br> [2] US EPA 2011 cited by INERIS (2010) <br> [3] Artola-Garicano, 2002 cited in EC (2008) <br> [4] EC (2008) <br> [5] EC (2008) <br> [6] Bayen et al. (2019) |
| Organic carbon adsorption coefficient (log K$_{OC}$) [a] | [1] 4.9 (4.5-5.2, exp., suspended matter/river water) <br> [2] 4.4 (suspended matter) <br> [3] 3.76-4.65 (exp. suspended matter, small streams with high input of waste water; 9.7-13.1 % OC) <br> [4] 4.0 (sediment) <br> [5] 3.04 – 4.23 (exp., sediment) <br> [6] 3.9 (K$_{oc}$ 8678 L/kg, est., EPIsuite v. 4.0) <br> [7] 3.41 (exp., HPLC) <br> [8] 3.7 - 4.13 (exp., soil desorption test) <br> [9] 4.47 (est. as log K$_{OC}$ = 0.81 x log K$_{OW}$ + 0.10) | [1] Winkler et al. (1998) <br> [2,4] Fooken (2004) cited in EC (2008) <br> [3,5] LfU-BW (2001) cited in EC (2008) <br> [6] US EPA 2011 cited by INERIS (2010) <br> [7, 8] Müller 2002 cited in EC (2008) <br> [9] See Appendix 1 |
| Sediment adsorption coefficient (K$_d$ [L/kg]) | [1] 9500 (2530 – 20100, exp. suspended matter, River Elbe, DE) <br> [2] 2900 (480-5580, exp. suspended matter, rivers and brooms, Hessen, DE) <br> [3] 600-4400 (exp. suspended matter, small streams with high input of waste water; 9.7-13.1 % OC) <br> [4] 12589 (501, 39811, 100000, exp. suspended matter, streams, US) <br> [5] 33 – 270 (exp. sediment, small streams with high input of waste water; 1.0-2.9 % OC) <br> [6] 144 (exp. sediment) | [1] Winkler et al. (1998) <br> [2] HLUG (2001) cited in EC (2008) <br> [3,5] LfU-BW (2001) cited in EC (2008) <br> [4] Standley et al. (2000) <br> [6] Fooken (2004) cited in EC (2008) |
| Aqueous hydrolysis DT$_{50}$ | cannot hydrolyze (OECD 111) | Van Ek, 2004 cited in EC (2008) |
| Aqueous photolysis DT$_{50}$ | [1] < 5 min (exp., water (not specified, laboratory grade), UV light (8-10 W, 245 nm)) | [1] Sanchez-Prado *et al.* (2004) |

| | [2] ~4 h (exp., lake water, 300-460 nm irradiation) | [2] Buerge *et al.* (2003) |
|---|---|---|
| Biodegradation in water DT$_{50}$ [d] [c] | [1] 150 (20 °C; based on exp. data on galaxolide/HHCB) [c] <br> [2] ~9 (exp., primary degradation[1], river water die-away test (river water mixed with activated sludge)) | [1] for risk assessment (EC 2008) <br> [2] Schafer and Koper (2006) cited in EC (2008) |
| Biodegradation in sediment DT$_{50}$ [d] | [1] 365 (12 °C; based on exp. data on galaxolide/HHCB) [c] | [1] for risk assessment (EC 2008) |
| Biodegradation in soil DT$_{50}$ [d] | [1] 365 (12 °C; based on exp. data on galaxolide/HHCB) [c] <br> [2] 92 (average; 50 ± 10 − 133 ± 40; soils amended with dried STP biosolids) | [1] for risk assessment (EC 2008) <br> [2] Chen et al. (2014) |

[a] Data obtained from HPLC-based or unknown methods are in grey font and were not used for EQS derivation. Data used for EQS derivation are in black front.

[b] The water-solubility of AHTN is independent of the pH. If the test item could dissociate, the solubility would increase. This does not take place in either alkaline or acidic aqueous media (pH=5, 7 or 9). The test item cannot dissociate in water due to a lack of relevant functional groups. Hence, the dissociation constant is irrelevant. Even if the test item was able to split in a cationic and anionic group, the water solubility of the substance is 1.25 mg/L and hence very low to obtain a suitable dissociation constant (EC 2008).

[c] For risk assessment (EC 2008) the biodegradation rate constants for surface water, sediment and soil were based both on the data for AHTN and on the results for HHCB. As a conservative approach, the rates for AHTN were taken as twice the rates for HHCB: 150 d in surface water (20 ºC) and 365 d in the soil and sediment compartments (12 ºC).

## 1.2    Regulation and environmental limits

AHTN was assessed under the OECD Cooperative Chemicals Assessment Programme (CoCAP) by the Netherlands, and assessed at the 28th Screening Information Dataset (SIDS) Initial Assessment Meeting (SIAM 28) in 2009. The SIDS Initial Assessment Profile (SIAP) states that AHTN may present a hazard for the environment (acute aquatic toxicity values below < 1 mg/L and not readily biodegradable) and that ecotoxicological data is available for information (OECD 2009).

AHTN is registered under REACH in the EU (ECHA 2012b). It was listed on the fourth Priority Substances List under the superseded Existing Substances Regulation (ESR) (ECHA 2007). The associated risk assessment concluded that for all environmental compartments no additional information and/or testing is needed, and no additional risk reduction measures are required (EC 2008). The lowest PNEC used for the sediment compartment was 1.72 mg/kg d.w. (5% OC) based on the lowest available NOEC of 7.1 mg/kg d.w. (growth, *Lumbriculus variegatus*, sediment with 2.06 % OC).

Importantly, AHTN is scheduled for re-evaluation in the Rolling Action Plan by the EU member state Germany for being a potential endocrine disruptor and due to high (aggregated) tonnage (EC 2020).

In Canada, AHTN was categorised as not Persistent (P), not Bioaccumulative (not B), and Inherently Toxic to the Environment (iT$_E$) by Environment Canada during the categorisation of the Domestic Substances List (DSL) and was not prioritised for further assessment (Environment Canada 2006).

Table 2 summarizes existing regulation and environmental limits in Switzerland, Europe and elsewhere for AHTN.

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[1] Primary biodegradation (biotransformation): The alteration in the chemical structure of a substance, brought about by biological action, resulting in the loss of a specific property of that substance (OECD definition).

*Table 2 Existing regulation and environmental limits for AHTN in Switzerland and Europe.*

| Europe | |
|---|---|
| EU Priority substance list | Inclusion in Priority substances list (ECHA 2007) |
| EU Community Rolling Action Plan | Scheduled for re-evaluation in 2020 (EC 2020) |
| REACH | Registered substance (ECHA 2012b) |
| **Switzerland** | |
| SR 817.02 Lebensmittel- und Gebrauchsgegenständeverordnung (LGV) | Restriction of concentrations in specific products |

## 1.3    Use and emissions

AHTN was identified as a High Production Volume (HPV; used at more than 1000 tonnes per annum in at least one member country or region) chemical by the OECD in 2004 (OECD 2004). According to the EU risk assessment report, AHTN was produced on one site in Europe, with a production volume 1000 - 5000 ton/y with circa 62% of the production volume being exported outside Europe (EC 2008). Information on whether AHTN is currently produced in Switzerland was not found. AHTN is the second highest volume polycyclic musk in terms of global production (following galaxolide/HHCB) with 247 tonnes used in Europe in 2004 (EC 2008). Emission of AHTN from production processes occurs via waste water treatment plants receiving process water containing organic waste and water containing aluminium chloride from the production plant in addition to AHTN (EC 2008). Polycyclic musks like AHTN are used as fragrances in cosmetics, detergents, fabric softeners, household cleaning products, air fresheners, and other products/applications. Based on these uses, it can be expected that most of the production volume will be released with waste water after use (Balk & Ford 1999a). Emission from landfill leachates has also been reported (Eggen et al. 2010).

## 1.4    Mode of action and sensitivity of taxonomic groups

AHTN is used as a fragrance, it confers an odour and is thus an organoleptic compound (Graham et al. 2013). Organoleptic compounds may be bioactive at ultra-trace levels.

A specific adverse mode of action has not been described to the best of our knowledge.

According to EC (2020), classification for genetic toxicity, carcinogenicity, toxicity to reproduction and neurotoxicity is not warranted in accordance with EU Classification, Labelling and Packaging of Substances and Mixtures (CLP) Regulation No. 1272/2008.

Adverse physiological effects have been reported at different levels of organization in aquatic and terrestrial organisms. Adverse physiological effects to early life stages of zebra fish were tested by Carlsson & Norrgren (2004), with the NOEC for lower heart rate being 10 µg/L. At the subcellular level, the activity of glutathione S-transferase and lipid peroxidation increased significantly at 50 µg/L, for catalase activity at 5 and 50 µg/L and glutathione reductase activity decreased significantly at 50 g/L (Blahova et al. 2018). No effect on body weight, body length, specific growth rate, and behaviour was observed at any of the tested concentrations (Blahova et al. 2018). In the earthworm *Eisenia fetida*, 28 d exposure (3-100 µg/g dry mass) via soil resulted in down-regulation of HSP70 gene expression and up-regulation of catalase gene expression at 30 µg/g and above, and up-regulation of calreticulin gene expressions at all tested concentrations (Chen et al. 2013).

AHTN is not listed in the most recent "Community Strategy for Endocrine Disrupters" as a substance with suspected or proven ED potential (EC 2007). AHTN has been assessed on its endocrine disrupting

potential as part of the human and environmental hazard assessment following REACH guidance and, based on a weight of evidence approach, AHTN is not endocrine disrupting.

However, AHTN is scheduled for re-evaluation in the Rolling Action Plan by the EU member state Germany for being a potential endocrine disruptor and due to high (aggregated) tonnage (EC 2020).

## 2 Environmental fate

Based on the Henry's law constants (Table 1) AHTN is moderately volatile[2].

### 2.1 Stability and degradation products

**Abiotic degradation**

AHTN was reported to be non-hydrolysable according to OECD guideline No. 111 (Van Ek, 2004 cited in EC (2008)).

Photodegradation by UV irradiation (low-pressure mercury lamps (8-10 W, 245 nm)) of AHTN in water (laboratory grade, not specified) was studied in 3 mL aliquots in synthetic quartz precision cells for 2–60 min (Sanchez-Prado et al. 2004). A half-life of less than 5 min was determined. After 30 minutes, more than 95% of AHTN was degraded.

Photodegradation by a wider light spectrum (mercury-vapour fluorescent lamps emitting UV-radiation between 300 and 460 nm with a maximum at 365 nm) in lake water from the Zürichsee (CH) and distilled water was studied in 25 mL aliquots of 1 µg/L AHTN (Buerge et al. 2003). Degradation of AHTN followed first-order kinetics with half-lives of approx. 4 h (photolysis rate constants were 4.6 $d^{-1}$ in lake water, 4.4 $d^{-1}$ in distilled water). Control experiments in the dark indicated that AHTN was not eliminated by other processes.

No data are available on biodegradation in sediment while several mineralisation and primary degradation studies are available for AHTN.

Mineralisation was marginal under standard test conditions for ready biodegradation. While three standard tests showed no oxidation and carbon dioxide formation, respectively (OECD 302C: modified MITI II, respirometric method (Rudio 1993); Modification of OECD 301B, Sealed vessel TIC test according to Birch and Fletcher, 1991 (King 1994); Modified Sturm test OECD 301B, CO2- evolution (Jenkins 1991 cited in EC (2008)), 21 % oxidation was observed after 3 weeks in the two-phase closed bottle test (NEN 6515, 1989 Oxygen uptake, Boersma and Hagens 1991, cited in EC (2008)). After repetitive additions of AHTN, a marginal oxygen consumption of 12 % was observed after 7 weeks.

Primary degradation was studied in a continuous activated sludge test with 10 µg/L $^{14}$C-AHTN and realistic STP operation conditions, total removal of the parent AHTN was 87.5 %, with 42.5 % removed by biotransformation and 44.3 % by sorption; volatilization played a minor role (3.3 %) (Lee et al. 2001, Federle et al. 2002; cited in EC (2008)).

In a die-away study for primary degradation with 5 and 50 µg/L $^{14}$C-AHTN in activated sludge, the half-life of the parent AHTN was 12-24 h and was largely biotransformed within 20 d. After 3 days, a variety of more polar metabolites were detected (Lee et al. 2001, cited in EC (2008)). In river water mixed with activated sludge (Schafer and Koper (2006) cited in EC (2008)), AHTN (5 µg/L) was rapidly transformed

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[2] The Henry's law constants in Table 1 translate into dimensionless Henry's law constants of 0.005 and 0.015, which according to the Rippen's classification (1990) indicates moderate volatilization (https://www.nicnas.gov.au/chemical-information/imap-assessments/imap-assessments/tier-ii-environment-assessments/environment-tier-ii-assessment-for-tonalide-and-related-polycyclic-musks)

to polar metabolites with only a small fraction of the theoretical oxygen demand having been incorporated. The overall half-life was approximately 9 days.

In a screening study on 64 soil samples, 28 % of these showed degradation of AHTN after 3 weeks, with an increasing number of positive samples with an increasing organic carbon (OC) content. Most of the 18 samples indicating degradation were sandy soils taken in forests, moors and at an industrial site (PFW 1997 cited in (EC 2008, HERA 2004, OSPAR Commission 2004)).

The dissipation of AHTN in sandy agricultural soils amended with sludge (sludge to soil ration representative of field conditions) was studied in a 1-year experiment with four different soils, with and without spiking of AHTN. The initial concentrations in spiked soil were 6 and 13 mg/kg soil, and between 0.1 and 0.27 mg/kg in non-spiked soil. The rates of degradation varied with the type of soil. In spiked soil, concentrations rapidly decreased during the first month and then decreased steadily in time. After 3 months, the concentrations had decreased to 65 to 80 % and remained during three months when soils were frozen. After one year, the concentrations in non-spiked soils ranged from 42 to 61 %, slightly higher variability was observed in spiked soils. The authors speculated that whether AHTN accumulates over repeated sludge applications may depend on the organic matter content of the soil. The leachates from spiked soils contained 0.04 to 0.18 % of the initial amount and non from non-spiked soils, both independent of organic matter content (DiFrancesco et al. 2004).

Dissipation of AHTN was investigated in soils subject to different biosolid amendments under field conditions in North China (Chen *et al.* 2014). Theoretical AHTN concentrations (based on the concentration measured in biosolids stock, assumed density of 1.3 g/cm$^3$) and actual concentrations in soils ranged from 10.8-86.2 µg/kg and 1.5-29 µg/kg, respectively. The average $DT_{50}$ was 92 d (50 ± 10 d – 133 ± 40 d).

As summarized in the EU risk assessment report (EC 2008), AHTN may be considered as "Inherently biodegradable, not fulfilling specific criteria" with a rate constant of 0 (see Table 6 in Part II of EC (2003)). The degradation rates for AHTN used for risk assessment were based on data on AHTN and the related galaxolide (HHCB) as twice the rates for HHCB: 150 d in surface water (20 °C) and 365 d in the soil and sediment compartments (12 °C) (EC 2008).

**Degradation pathway and products**

The minimal differences between the photolysis rate constants determined in lake water and distilled water indicate that AHTN is degraded primarily via direct photolysis and that indirect photochemical degradation by reactive oxygen species is of minor importance (Buerge et al. 2003).

The tentatively identified biodegradation products show that AHTN is not attacked on the hexamethyltetralin group but on the acetyl group via three different pathways: by formation of a five-ring with aldehyde group, by removal of the oxygen moiety or by addition of an oxygen (Sanchez-Prado et al. 2004).

More detailed information was not available for review, among others due to the original reports not being publically available.

## 2.2 Sorption/desorption processes

Table 1 and Appendix 1 presents available adsorption and partitioning coefficients for AHTN. Several adsorption coefficients have been reported from field studies ranging from 480 to 100000 L/kg for freshwater suspended matter and from 33 to 270 L/kg for sediment. Log $K_{oc}$ values range between 3.76 and 5.2.

Field derived partition coefficients in stream water samples taken from River Elbe in Germany varied in time from 2533 L/kg to 20773 L/kg, with average 9204 L/kg (Winkler et al. 1998). Normalization by organic carbon reduced the variation to some extent but correlation of $K_d$ and TOC was also not apparent: the mean field derived log $K_{oc}$ for the Elbe was 4.9 (range 4.5 – 5.2). It was concluded that sorption varies over time with the composition and the quantity of the suspended materials.

The information available for sewage sludge is more abundant than for suspended matter and sediment.

In a study to determine $K_d$ as the ratio between the concentration of AHTN in sludge and in effluent, the relation between 60 effluent and sludge concentrations (from waste water treatment plants in Italy, Spain and Greece) was expressed as regression line with a slope of 45° resulting in log $K_d$ = 3.95 (3.47 and 4.36). The deviation of the best fitting line from the 45° line was analysed and found to be significant, indicating that $K_d$ was not independent of AHTN concentration (Blok 2005 cited in EC (2008)). Although it has not been used here for sediment EQS derivation, it provides useful information on sorption/desorption of AHTN.

Similarly, another study showed the deviation from a constant partition coefficient, implying that at low AHTN concentration the effluent concentration was relatively high (small $K_d$) and at high AHTN concentration the effluent concentration was relatively small (high $K_d$). This indicates, that sludge has a higher adsorptive potential at higher AHTN concentrations (Van der Hoeven (2005) cited in EC (2008)).

Adsorption coefficients for soils varies within the same range as that reported for suspended matter and sediments, with log $K_{oc}$ from 3.7 to 4.13 (Müller 2002 cited in EC 2008). Given that there are values for sediments and suspended matter, soil values have not been used here for EQS derivation.

## 2.3 Bioavailability

Bioavailability is a complex process which depends on many factors including the sorption capacity of the sediment considered (e.g. OC content), the hydrophobicity of the compound, and the physiology, feeding behaviour and burrowing activity of the benthic organism considered (Warren et al. 2003).

The scientific opinion of the EFSA on the effect assessment for pesticides on sediment organisms recognizes that "*the most appropriate metric for bioavailability in soils and sediments appears to be the 'freely dissolved pore water concentration' rather than the total sediment concentration, particularly for compounds with a log $K_{OW}$ < 5*" (EFSA 2015).

No studies on the bioavailability of AHTN were available based on the literature research performed for this dossier. Studies on $K_{oc}/K_d$ of AHTN (section 2.2) indicate that sorption to organic carbon and thus bioavailability in different compartments depends on AHTN concentration and the type of organic carbon present, rather than the concentration of organic carbon alone.

## 2.4 Bioaccumulation and biomagnification

Two reliable OECD 305E studies (Bioaccumulation: Flow-through Fish Test) are available for AHTM. From a 28 day exposure to radiolabelled AHTN of *Lepomis macrochirus*, bioconcentration factor (BCF) values based on concentrations of parent compound averaged 258±1, 845±28, and 597±1 for edibles, non-edibles, and whole fish, respectively. In both exposure water and fish and fish parts, analyses of radioactivity detected mostly parent compound plus one other metabolite (Van Dijk 1996 cited in EC 2008). AHTN was rapidly transformed into more polar and water soluble (low log $k_{ow}$) metabolites that were readily excreted (38-50 % per day). A second study available for *Danio rerio* (formerly

*Brachydanio rerio,* Butte 1999 cited in EC 2008) with a 14 d uptake exposure report a BCF for AHTN of ca. 600 (whole fish) L/kg is based on fresh weight.

A calculated BCF of 697 has been reported (EPISuite 4.0; (INERIS 2010)).

AHTN is regularly detected in fish and mussels/oysters, but is not ubiquitous in tissue samples. Reported tissue concentrations in fish liver (*Cyprinus carpio, Abramis brama, Sander lucioperca*) at the Po River were < 1 µg/kg d.w. (LOD) (Luigi et al. 2015), <1–32.8 µg/kg lipid weight in whole body (n = 30, *Anguilla rostrata, Micropterus dolomieu, Micropterus salmoides, Morone americana, Ameiurus nebulosus*, *Ictalurus punctatus* and *Ameiurus catus)*, in the Hudson River, USA (Reiner & Kannan 2011), and 2.69 µg/kg lipid weight average whole body of different species and water ways in Berlin, DE (max. 88.3 µg/kg; n = 324; 97 samples < LOQ) (Heberer et al. 2004). Mussel/oyster tissue have reported concentrations ranging from e.g. 42.2–65.9 µg/kg lipid weight (zebra mussel, n = 4) (Reiner & Kannan 2011) and <LOD–360 µg/kg lipid weight (*Mytilus coruscus, Mytilus edulis, Crassostrea gigas*; Nakdong River) (Lee et al. 2014). Reported field BCF range from 597 in bluegill sunfish (*Lepomis macrochirus*; River Ruhr; reported as BCF but not excluding dietary uptake) (Balk & Ford 1999a) to 1069 in eels (various streams in Berlin; reported as BCF but not excluding dietary uptake) (Fromme et al. 2001).

Thus the BCF of AHTN in standard OECD bioaccumulation studies were below the Bioaccumulation criterion of 2000, and this is confirmed by the results of field studies (EC 2008).

Concerning the risk of benthic invertebrates to transfer toxic and bioaccumulative substances to higher trophic levels, the EFSA scientific opinion for sediment risk assessment proposes to perform spiked sediment bioaccumulation tests with benthic invertebrates for substances that show significant bioaccumulation in fish (BCF ≥ 2000) when the substance is (1) persistent in sediment (DT$_{50}$ >120 d in water-sediment fate studies) and log K$_{ow}$ >3; or (2) non-persistent in sediment, log K$_{ow}$ >3 and >10 % of the substance found in the sediment in a water-sediment fate study (EFSA 2015). The bioaccumulation criterion (BCF ≥ 2000) is not fulfilled, and studies on the bioaccumulation of AHTN in spiked sediments are not available.

There are, however, bioconcentration studies with midge larvae (*Chironomus riparius*) and the worm *Lumbriculus variegatus* exposed to AHTN in water for 12 days, resulting in a BCF = 50-112 and 6918, respectively at steady-state (Artola-Garicano et al. 2003). Comparison of BCF and the study of the cytochrome P-450 inhibitor piperonyl butoxide indicates that *C. riparius* is able to transform AHTN while *L. variegatus* is not.

In a Mangrove system in China, the average field BSAF (normalized for lipid contents and OC content of the sediments) in caged mussels (*Perna viridis*) was 3.2 (n = 3) and 7.9 in clams (*Polymesoda expansa*) collected on site (n = 6) (Bayen et al. 2019). These results show that AHTN is accumulated in bivalves living in close contact with sediments.

The EU Risk Assessment Report for AHTN (EC 2008) also reviewed field data on biomagnification. In seven field studies on food chain accumulation including aquatic mammals and sharks as well as water birds around the northern hemisphere, AHTN showed low concentrations in fish and mussel tissues (range 0.1 mg/kg lipids) and at or below the detection limits in predators. It was concluded that AHTN did not show increasing concentrations at higher trophic levels so biomagnification along the food chain does not occur.

Based on available data, it is concluded that benthic invertebrates probably do not contribute to the risk to higher organisms through trophic transfer. Studies on sediment bioaccumulation in deposit feeders were not available to the best of our knowledge. Bioaccumulation based on this feeding strategy can thus not be assessed.

# 3  Analysis

## 3.1  Methods for analysis and quantification limit

An overview of general challenges in quantification of AHTN and reported techniques has been provided by Bester (2009). Detection limits of approx. 0.15 µg/kg d.w. are achieved by Soxhlet extraction followed by clean-up with column chromatography and quantification with gas chromatography-mass spectrometry GC-MS) (Peck et al. 2006). Use of quadrupole MS-MS coupling did not improve sensitivity of the method.

*Table 3 Methods for AHTN analysis in sediments and corresponding limits of detection (LOD) and limits of quantification (LOQ) (µg/kg d.w.). n. a. means not reported.*

| LOD | LOQ | Analytical method | Reference |
|---|---|---|---|
| 0.15 | n.a. | Soxhlet extraction, Column chromatography, GC-MS and GC-MS-MS | Peck et al. (2006) |
| 5 | n.a. | GC-MS-MS | Oniris, LABERCA (Ecotox Centre, unpubl. data) |
| 0.122 | 0.407 | In-cell clean-up pressurized liquid extraction, GC-MS-MS | Pintado-Herrera et al. (2016) |

## 3.2  Environmental concentrations

Measured environmental concentrations (MEC) of AHTN in sediments are summarized in Table 4. MECs in sediments from Switzerland are available for small streams. Concentrations of AHTN quantified in sediments from 5 streams mainly affected by agricultural practices in 2017 ranged from <0.122 µg/kg d.w. to 2.10 µg/kg d.w. (Table 4). A similar range of concentrations was reported in suspended particulate matter at these same streams. In a monitoring study performed in August 2018 at 18 small streams with different pollution sources concentrations of AHTN ranged from < 5 µg/kg d.w. to 7.52 µg/kg d.w. in the <2 mm sediment fraction and from <5 µg/kg d.w. to 20.43 µg/kg d.w. in the <63 µm fraction.

AHTN concentrations are also available for sediments from rivers in European countries. A recent study reported concentrations in surface sediments from the Po River ranging from 16 µg/kg d.w. to 159 µg/kg d.w. (Viganò et al. 2015). Decreasing concentrations of AHTN in surface water bodies were attributed to decreasing use (Viganò et al. 2015). Concentrations in three surface water bodies in Austria receiving waste water treatment plant effluents ranged between < 10 µg/kg d.w. to 20 µg/kg d.w. (Clara et al. 2011). Concentrations in the same order of magnitude were reported at the River Lippe, Germany in campaigns performed in 2000 and 2001 while AHTN concentrations reached 1399 µg/kg d.w. in 1999 in the area near the river mouth, attributed to the accumulation of highly contaminated particulate matter from upstream areas (Dsikowitzky et al. 2002; Kronimus et al. 2004). AHTN concentrations were also in the range of several hundred µg/kg d.w. in sediments from China

rivers (Hu et al. 2011) and in sediments collected in 2006 in the Hudson River, USA (Reiner & Kannan 2011).

Concentrations of AHTN in sediments from Lake Ontario, Canada were 0.96 µg/kg d.w. in 2003 (Peck et al. 2006) and up to 2.3 µg/kg d.w. in coastal monitoring sites close to the Nakdong River (Lee et al. 2014). In coastal sediments from the Adriatic Sea, Italy concentrations ranged from 0.6 µg/kg d.w. to 24.3 µg/kg d.w. (Combi et al. 2016) while in mangrove sediments from Singapore AHTN ranged from <0.6 µg/kg d.w. to 47 µg/kg d.w. (Bayen et al. 2019).

*Table 4 Measured environmental concentrations (MEC) of AHTN in Switzerland, Europe and elsewhere around the world. All concentrations expressed as µg/kg d.w. for sediment if not indicated. n.d. not detected*

| Country | MEC (min-max) | No. of sites | Comments | Reference |
|---|---|---|---|---|
| Switzerland | <0.122-2.10 (sediment, fraction < 2 mm, detected at all sites) <0.122-2.42 (suspended particulate matter) | 5 | Top sediment layer, small streams mainly affected by agricultural activities, samples monthly from March to October 2017 | Ecotox Centre, unpubl. data |
| Switzerland | <5-7.52 (sediment, fraction < 2 mm) <5-20.43 µg/kg d.w. (sediment, <63 µm fraction) | 18 | Top sediment layer, small streams, August 2018 | Ecotox Centre, unpubl. data |
| Austria | <10 and 20 | 3 | Surface waters receiving treated wastewater, grab samples | Clara et al. (2011) |
| Germany | <0.5 – 23 (Feb. 2000) <0.5 – 29 (Aug. 2000) n.d. – 90 (Mar. 2001) | 9 | Lippe River, no information on sediment sampling | Kronimus et al. (2004) |
| Germany | 2 – 1399 (1999) | 19 | Lippe River, top 0-5 cm | Dsikowitzky et al. (2002) |
| Italy | 16 - 159 | 5 | River Po, top 0–10 cm of bed sediment in natural deposition areas (depth ≥1 m) | Luigi et al. (2015) |
| China | <0.33 - 300.4 | 13 | Haihe River, Dagu Drainage River, and Chentaizi Drainage River, no information on sediment sampling | Hu et al. (2011) |
| USA | 113–544 (2006) | 3 | Hudson River (New York State), top 0-5 cm | Reiner & Kannan (2011) |
| Lake Ontario, Lake Erie | 0.96 (2003) | 2 | Top 2 cm; concentrations in a similar range since 1940 | Peck (2006) |
| Singapore | < 0.6 - 47 ± 10 | 12 | Mangrove sediments, 2 samples per site | Bayen et al. (2019) |
| Korea | ND–2.3 (2009) ND–1.0 (2010) | 22 25 | Nakdong River Coastal monitoring sites, top 0-10 cm | Lee et al. (2014) |
| Adriatic Sea | Northern Adriatic 1.3 - 24.3 Central Adriatic 0.9 - 13.6 Southern Adriatic 0.6 - 13.4 | 15 17 15 (based on Fig. 3) | Top 0.5 cm of undisturbed sediment ; mean OC: Northern Adriatic 1% Central Adriatic 0.7% Southern Adriatic 0.7% | Combi et al. (2016) |

## 4 Effect data (spiked sediment toxicity tests)

A non-filtered bibliographic search was performed for AHTN (by CAS numbers) in the US Ecotox Data Base (U.S. EPA 2016) which did not yield data on sediment organisms. Likewise, a search in the German Environmental Office database ETOX did not yield any relevant results. A key word search performed on Scopus (AHTN OR tonalid OR tonalide OR 6-acetyl-1,1,2,4,4,7-hexamethyltetraline OR musk + sediment + toxicity, no restriction regarding publication date) resulted in 216 publications, none of which were based on spiked sediment tests. Potentially unpublished data was searched for in registration information, risk assessment dossiers and EQS dossiers (Balk & Ford 1999b, EC 2008, ECHA 2012a, HERA 2004, OSPAR Commission 2004). The EU risk assessment report on AHTN lists three toxicity tests performed with benthic organisms exposed via spiked sediment (EC 2008). The original reports are not available but were performed under GLP and are completely documented as indicated in EC (2008; p. 106). Thus assessment of relevance and reliability of the studies was performed based on available information.

Relevance ("C" score in the table below) and reliability ("R" score in the table below) of studies are evaluated according to the CRED-criteria (Moermond et al. 2016) adapted for sediments (Casado-Martinez et al. 2017).

According to the EU TGD (EC 2018) "*What is considered chronic or acute is very much dependent on 1) the species considered and 2) the studied endpoint and reported criterion*". According to EFSA, true chronic tests should cover a range of 28-65 d when half-life of a pesticide in sediment is >10 d (EFSA, 2015). All available data originate from 28 d tests and are thus considered as "chronic" endpoints.

Table 5 Sediment effect data collection for AHTN in mg/kg d.w. Data were evaluated for relevance and reliability according to the CRED criteria for sediments (Casado-Martinez et al. 2017). Data assessed as not relevant and not reliable is in grey font. Data used for QS development is underlined. Abbreviations: n. a. = not available.

| Group | Species[a] | Test compound | Exposure | Equilibration time | Endpoint | Test duration | Effect concentration | Value [mg/kg d.w.] | Sediment type | Normalized value [mg/kg d.w., 1 %OC] | Normalized value [mg/kg d.w., 5 %OC] | Chem. analysis | Note | Validity | References |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Acute toxicity data in freshwater |||||||||||||||| 
| No data available |||||||||||||||| 
| Acute toxicity data in marine water |||||||||||||||| 
| No data available |||||||||||||||| 
| Chronic toxicity data in freshwater |||||||||||||||| 
| Insecta | *Chironomus riparius* | AHTN | static | 7 d | Development rate | 28 d | NOEC | 101 | OECD 218 artificial sediment (5% peat, 7 5% quartz sand, 20% kaolinite clay, 0.05% CaCO₃, 0.2-0.25% Urtica powder pH 6.5 -7.0, TOC 2.4% | 42.1 | 210.4 | Measured | | R1/C1 | Egeler & Gilberg (2004a) cited in (EC 2008, 2020) |
| Insecta | *Chironomus riparius* | AHTN | static | 7 d | Emergence ratio | 28 d | NOEC | 100 | " | 41.7 | 208.3 | Measured | NOEC for males and females taking recovery into account[a] | R1/C1 | Egeler & Gilberg (2004a) cited in (2020) |
| Crustacrea | *Hyalella azteca* | AHTN | static | 7 d | Survival | 28 d | NOEC | 18.2 | OECD 218 artificial sediment (5% peat, 75% quartz sand, 20% kaolinite clay, 0.05% CaCO₃, 0.2-0.25 % Urtica powder pH 6.5 -7.1, TOC 2.15%) | 13.5 | 67.4 | Measured | NOEC from measured concentration on day 0; concentration stable during the experiment | R1/C1 | Egeler (2004) cited in (EC 2008, 2020) |
| Crustacrea | *Hyalella azteca* | AHTN | static | 7 d | Growth rate | 28 d | NOEC | 18.2 | " | 8.47 | 42.3 | Measured | " | R1/C1 | Egeler (2004) cited in (EC 2008, 2020) |
| Oligocheta | *Lumbriculus variegatus* | AHTN | static | 7 d | Survival | 28 d | NOEC | >79 | OECD 218 artificial sediment (5% peat, 75% quartz sand, 20% kaolinite clay, 0.05% CaCO₃, 0.2-0.25 % Urtica powder pH 6.5 -7.1, TOC 2.06%) | >38.3 | >191.7 | Measured | NOEC from measured concentration on day 0 | R1/C1 | Egeler & Gilberg (2004b) cited in (EC 2008, 2020) |
| Oligocheta | *Lumbriculus variegatus* | AHTN | static | 7 d | Reproduction | 28 d | NOEC | 16.43 | " | 7.98 | 39.9 | Measured | NOEC from measured concentration on day 0 (62% of 26.5 mg/kg d.w. nominal) | R1/C1 | Egeler & Gilberg (2004b) cited in (EC 2008, 2020) |

| Group | Species[a] | Test compound | Exposure | Equilibration time | Endpoint | Test duration | Effect concentration | Value [mg/kg d.w.] | Sediment type | Normalized value [mg/kg d.w., 1 %OC] | Normalized value [mg/kg d.w., 5 %OC] | Chem. analysis | Note | Validity | References |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Oligocheta | *Lumbriculus variegatus* | AHTN | static | 7 d | Biomass | 28 d | NOEC | <u>7.1</u> | " | 3.45 | 17.2 | Measured | NOEC derived from measured concentration on day 0 (62% of 11.5 mg/kg d.w. nominal) | R1/C1 | Egeler & Gilberg (2004b) cited in (EC 2008, 2020) |
| Oligocheta | *Lumbriculus variegatus* | AHTN | static | 7 d | Reproduction | 28 d | NOEC | 50 | 5% peat, 75% quartz sand, 20% kaolinite clay, 0.05% CaCO$_3$, 0.2-0.25 % Urtica powder pH 6.5 -7.1, TOC 3.2% | 15.6 | 78.1 | Nominal | Number of non-divided adults, worms with no new end; effects related to lack of burrowing and starvation | R3/C1 | Liebig (2005) |
| **Chronic toxicity data in marine water** | | | | | | | | | | | | | | | |
| No data available | | | | | | | | | | | | | | | |

[a] EC (2008) reports NOEC emergence of 500 mg/kg d.w. (nominal).

## 4.1 Graphic representation of effect data

All relevant and reliable data have been plotted without further normalization due to the little difference among tests and data (Figures 2).

The NOEC of biomass of oligochaetes (*L. variegatus*, 7.1 mg/kg d.w.) is the lowest NOEC followed by biomass of crustaceans (*H. azteca*, 18.2 mg/kg d.w.) and emergence ratio of insect larvae (*C. riparius*, 100 mg/kg d.w.).

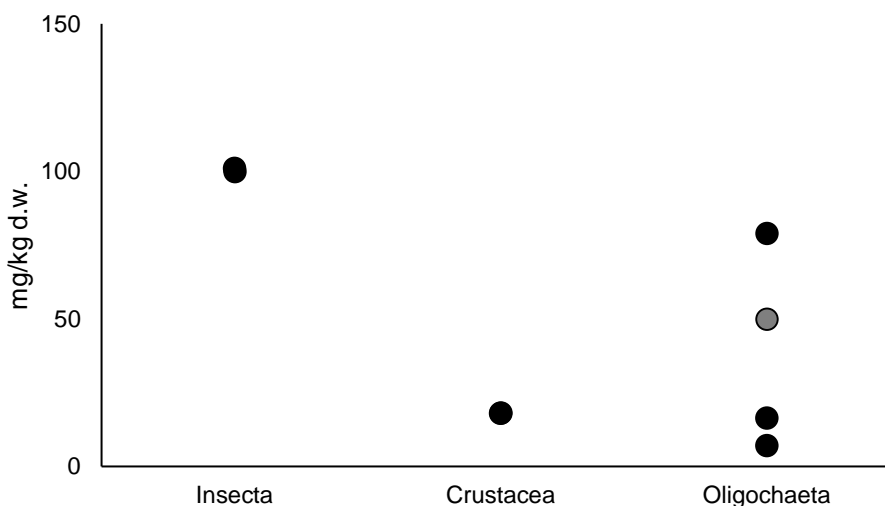In the absence of acute effect data, no ratio of relevant acute to chronic data can be derived*.*



*Figure 1 Graphical representation of chronic effect data from spiked sediment toxicity tests with AHTN. Data are not normalized for OC.*

## 4.2 Comparison between marine and freshwater species

No marine effect data were available for AHTN.

## 4.3 Overview of the most sensitive relevant and reliable long-term study

According to the EC EQS TGD (EC (2018) p. 25): "*All available data for any taxonomic group or species should be considered, provided the data meet quality requirements for relevance and reliability*".

The chronic effect data for *Lumbriculus variegatus* (survival, reproduction, biomass), *Chironomus riparius* (emergence ratio, development rate), and *Hyalella azteca* (survival, biomass) have been evaluated as R1/C1 based on available information in EC (2008) and EC (2020). The original reports were not available for evaluation but the information reported is detailed and the data were considered reliable without restrictions according to the data quality assurance control (EC 2008, 2020). In the following, information on the critical study on *Lumbriculus variegatus* (survival, reproduction, biomass) is summarized.

Egeler & Gilberg (2004b) "AHTN/Tonalide: A study on the toxicity to the Aquatic Oligochaete *Lumbriculus variegatus*. ECT Study Number: AE1LA. ECT Oekotoxikologie GmbH, Germany. Report to PFW Aroma Chemicals." as cited in (EC 2008 and EC 2020).

- Species: *Lumbriculus variegatus*

- Guideline: OECD TG 218 (Draft December 2002), tests were carried out under GLP and were reported to having been completely documented.
- Origin: test organisms obtained from cultures at ECT Oekotoxikologie GmbH since Jan. 1998. The animals were originally obtained from Fischfutter Etzbach (D-53894 Mechernich-Bergheim, Germany). The species identity of the cultured organisms was confirmed according to Brinkhurst (1971). The test animals were 'synchronised' before the start of the test to avoid high variation in the test results according to OECD 223.
- Experimental sediment: the sediment contained 5 % Sphagnum moss peat, 75 % quartz sand (>50 % in range 50-200 μm), 20 % kaolinite clay and 0.05 % calcium carbonate to adjust the pH between 6.5 and 7.1. The organic carbon content was 2.06 %. 0.2- 0.25 % Urtica powder was added as food. The formulated sediment was conditioned for 7 days prior to application of the test material.
- Spiking and equilibration time: AHTN was solved in acetone to prepare the stock solutions for each concentration. The proper volumes were mixed first with dry quartz sand allowing the solvent to evaporate. The sand was then mixed with the conditioned sediment to achieve the intended nominal concentration levels. Each glass vessel contained a layer of 1.5 to 3 cm of sediment and the overlying water volume was 3.5 to 4.5 times the sediment volume. Both a control and a solvent control were included. The test animals were introduced after an equilibration period of 1 week.
- Overlying water: Elendt medium M4 was used as overlying water and was slightly aerated during the test. Results of water quality measurements (water hardness, temperature, pH, dissolved oxygen and ammonia) reported and within acceptable values.
- Bioassays: the tests were carried out at 20 °C; light/dark cycle was 16 h light, 8 h dark; light intensity was 400 to 600 lux. No additional food was given during the test. *Lumbriculus variegatus* was tested in concentrations ranging from 5 to 140 mg/kg d.w. with step size 2.3. Four replicates with 10 regenerated animals were used per test concentration and in the solvent control, whereas six replicates were used in the control.
- Test endpoints: survival, reproduction, biomass measured after 28 d.
- Measured AHTN concentrations: samples of porewater and overlying water were extracted by SPE using Speedisks. Sediment samples were freeze-dried and analysed by GC/MS after solvent extraction. The results are reported by Belfroid and Balk (2005; as cited in EC (2008) and EC (2020)).The test concentrations were measured on day 0, 9, 20 and 28 in the control, 26.5 and 140 mg/kg d.w. At the start of the test, sediment concentration was on average 62% of the intended nominal test concentration. At termination of the test, the average concentration in sediment was 91% of the concentration at the start. Concentrations are reported on measured concentrations at start of the test.
- Statistics: normal distribution of data was tested with the Kolmogorov-Smirnov goodness-of-fit test (two sided, $p \leq 0.05$). Cochran´s test (two-sided, $p \leq 0.05$) was used to test variance homogeneity. One-way Analysis of Variance (ANOVA) and subsequent multi-comparison tests (Dunnett's Test; one-sided, smaller) were used to calculate whether there were significant differences ($p \leq 0.05$) between the controls and the various test item concentrations. To determine significant differences between the controls (control and solvent control) the replicates of each control were tested for normal distribution homogeneity (Kolmogorov-Smirnov-test) and for homogeneity of variances (Cochran´s test); thereafter a pair wise comparison test (Student-t Test or Welch T-test) were used. If these tests detected no significant differences, all control and solvent control replicates were pooled.

- Results: survival was not affected up to the highest test concentration. Based on nominal concentrations, reproduction (total number of worms, including adult and regenerated worms) was significantly inhibited (43 %) at 61 mg/kg d.w., whereas the inhibition was 13 % at 26.5 mg/kg (= NOEC$_{repr.}$). The EC$_{50repr.}$ was 87.2 (65.1 – 128.9) mg/kg d.w., the EC$_{15repr.}$ was 26.4 (11 – 39.1) mg/kg d.w. Growth as measured by biomass was significantly inhibited (20 %) at 26.5 mg/kg d.w., whereas at 11.5 mg/kg d.w. the inhibition was 13% (= NOEC$_{biomass}$). The EC$_{50biomass}$ was 120.6 (81.1 – 254.2) mg/kg d.w., the EC$_{15biomass}$ was 19.4 (6.5 – 31.6) mg/kg d.w. The lowest NOEC was 11.5 mg/kg d.w. As the actual concentration on day 0 was not measured at the level of the NOEC, the actual concentration was determined based on the mean measured concentration: 62% of 11.5 mg/kg d.w. equalling 7.13 mg/kg d.w. The overall NOEC was thus reported as 7.1 mg/kg d.w. (measured concentration).

# 5 Derivation of QS$_{sed}$

According to the EC TGD for EQS, sediment toxicity tests, aquatic toxicity tests in conjunction with equilibrium partitioning (EqP) and field/mesocosm studies are used as several lines of evidence to derive QS$_{sed}$ (EC 2018). Thus, in the following, the appropriateness of the deterministic approach (AF-Method), the probabilistic approach (SSD method) and the EqP approach were examined.

## 5.1 Derivation of QS$_{sed, AF}$ using the Assessment Factor (AF) method

The derivation of QS$_{sed, AF}$ is determined using assessment factors (AFs) applied to the lowest credible datum from long-term toxicity tests.

The lowest long-term effect datum available for AHTN is the NOEC of 7.1 mg/kg d.w. or 344.66 mg/kg-OC (2.06 % OC, Table 6) for the biomass of *Lumbriculus variegatus*.

*Table 6 Most sensitive relevant and reliable chronic data summarized from Table 5.*

| Species | Exposure duration [d] | Endpoint | NOEC/EC$_{10}$ [mg/kg d.w.] | OC [%] |
|---|---|---|---|---|
| *Chironomus riparius* | 28 d | Emergence ratio | 100 | 2.4 |
| *Hyalella azteca* | 28 d | Total biomass | 18.2 | 2.15 |
| *Lumbriculus variegatus* | 28 d | Biomass | <u>7.1</u> | 2.06 |

In case of long term tests (NOEC or EC$_{10}$) being available for three species representing different living and feeding conditions, the TGD recommends the application of an assessment factor of 10 on the lowest credible datum (Table 11 in EC (2018)).

The most sensitive datum (NOEC) considers the biomass of individuals *L. variegatus* and was lower than the NOEC for reproduction (number of individuals) (Table 5).

Although AHTN is scheduled for re-evaluation in the Rolling Action Plan by the EU member state Germany for being a potential endocrine disruptor and due to high (aggregated) tonnage (EC 2020), according to available data there is no evidence of ED potential and AHTN is not listed in the Community Strategy for Endocrine Disrupters (EC 2007) as a substance with suspected or proven ED potential (EC 2008). There is currently no evidence that fragrances exert relevant effects not covered by the endpoints listed here. The suggested assessment factor is 10 in accordance with TGD 27:

$$QS_{sed,AF} = \frac{lowest\ EC10\ or\ NOEC}{AF}$$

$$QS_{sed,AF} = \frac{344.66\left(\frac{mg}{kg-OC}\right)}{10} = 34.466\left(\frac{mg}{kg-OC}\right)$$

The application of an AF of 10 to the lowest credible chronic datum results in a QS$_{sed,AF}$ = 34.47 mg/kg-OC, which corresponds to 1.72 mg/kg d.w. for a sediment with 5 % OC or 0.344 mg/kg d.w. for a sediment with 1 % OC representing a worst case scenario in Switzerland.

## 5.2 Derivation of QS$_{sed,SSD}$ using the species sensitivity distribution (SSD) method

The minimum data requirements recommended for the application of the SSD approach for EQS water derivation is preferably more than 15, but at least 10 NOECs/EC$_{10}$S, from different species covering at least eight taxonomic groups (EC (2018), p. 43). In this case, not enough data from spiked sediment toxicity tests are available for applying the SSD approach.

# 6 Derivation of QS$_{sed,EqP}$ using the Equilibrium Partitioning approach

If no reliable sediment toxicity data are available, the Equilibrium Partitioning (EqP) can be used to estimate the EQS$_{sed,EqP}$. This approach, developed for non-ionic substances, is used here for comparison purposes given the small data base of sediment toxicity studies.

## 6.1 Selection of QS for water

For the derivation of the EQS$_{sed,EqP}$, a PNEC for the aquatic freshwater environment derived with a methodology similar to the procedure described in the TGD for deriving the AA-EQS for freshwater (e.g. with regard to the AF) should be used.

A water quality standard for AHTN was not found, whereas a PNEC for freshwaters was derived by Balk & Ford (1999b), the European Union Risk Assessment Report (EC 2008), and in the Australian "Tonalide and related polycyclic musks: Environment tier II assessment" (Australian Government 2016). Additionally, a PNEC for marine waters was derived in the "OSPAR Background Document on Musk Xylene and Other Musks" (OSPAR Commission 2004).

The most recent PNEC value used for the EU risk assessment (EC 2008) was set at 2.8 µg/L, derived from the EC$_{10}$ for the marine copepod *Acartia tonsa* and an assessment factor of 10. Balk & Ford (1999b) derived a PNEC of 3.5 µg/L from a NOEC of 0.035 mg/L from a 35-day early life stage tests with fathead minnows (*Pimephales promelas*) combined with an assessment factor of 10. The lowest PNEC for freshwater of 0.35 µg/L was derived for Australia and is based on the lowest available chronic endpoint (34-d fish toxicity test with zebrafish) and an assessment factor of 100. According to the assessment by the Australian Government, a conservative assessment factor of 100 was selected as - although chronic ecotoxicity data were available across three aquatic trophic levels - AHTN was considered to have moderate bioaccumulation potential in fish. The same PNEC was derived by Haskoning 2003 (cited in OSPAR Commission (2004)) and was used for risk assessment for marine aquatic organisms (OSPAR Commission 2004).

The most relevant PNEC for the derivation of a EQS$_{sed,EqP}$ is the PNEC of 2.8 µg/L (EC 2008).

## 6.2    Selection of partition coefficient

One of the main factors influencing the application of the EqP model is the choice of the partition coefficient. It is stipulated in the ECHA 2017 guideline (p. 143, ECHA (2017)) that "To increase the reliability of PNEC sediment screen derived using the EqP, it is imperative that a conservative but realistic partitioning coefficient (e.g. $K_d$, $K_{oc}$, $K_{ow}$) is chosen. A clear justification must be given for the chosen coefficient and any uncertainty should be described in a transparent way."

The EC EQS TGD requires deriving a geometric mean of all available $K_{oc}$ values including one derived from a log $K_{ow}$ value (EC 2018).

Reported experimental log $K_{oc}$ for AHTN range from 3.76 to 5.2. A log $K_{oc}$ value of 4.47 is estimated from the log $K_{ow}$ of 5.40. A geomean of 4.36 (see Appendix 1) is used for EQS derivation via EqP as listed in Appendix I.

## 6.3    Selection of OC content for a reference sediment

To account for the influence of OC content on QS$_{sed,EqP}$ development, calculations have been performed for a standard sediment according to the EU TGD with 5 % OC (EC 2018). As 5 % OC might not be representative for sediment in Switzerland, calculation was made as well for a worst case scenario considering measurement on total sediment with 1 % OC (approx. 10th percentile of OC content in Swiss Rivers).

## 6.4    Derivation of QS$_{sed,EqP}$

For the derivation of QS$_{sed,EqP}$, the partition coefficient between water and sediment has been estimated as the fraction of organic carbon multiplied by organic carbon partition coefficient ($Kp=f_{OC}*K_{OC}$) as proposed by Di Toro et al. (1991) for non-ionic organic chemicals (Table 7). The authors considered that, for sediment with an organic fraction higher than 0.2 %, organic carbon is the main driver for chemical sorption.

An additional AF of 10 should be applied to the resulting QS$_{sed,EqP}$ for substances with log $K_{ow}$ >5. Reported experimental log $K_{ow}$ for AHTN range from 5.4-5.7, with a geometric mean of 5.55 (Table 1), estimated log $K_{ow}$ range from 5.32-6.35. Thus, application of the additional AF is warranted and the resulting QS$_{sed,EqP}$ are 65.1 µg/kg d.w. for sediments with 1 % OC and 323.6 µg/kg d.w. for sediments with 5 % OC when using a PNEC$_{water}$ of 2.8 µg/L.

Table 7 Derived QS$_{sed,EqP}$ for a mean $K_{OC}$ based on Appendix I and the PNEC for water derived by (EC 2008). The partition coefficient solid-water sediment (Kp$_{sed}$) is estimated for a sediment with 5% OC (standard EC TGD sediment) and 1% TOC (worst case scenario in Switzerland).

| OC [%] | $K_{oc}$ [L/kg] | Kp$_{sed}$ [L/kg] | K$_{sed-water}$ [m³/m³] | PNEC$_{freshwater}$ [µg/L] | QS$_{sed,EqP}$ [µg/kg w.w.] | QS$_{sed,EqP}$ [µg/kg d.w.] | Included additional AF |
|---|---|---|---|---|---|---|---|
| 1 | 23085 | 230.85 | 116.22 | 2.8 | 250.33 | **65.09** | 10 |
| 5 | 23085 | 1154.25 | 577.92 | 2.8 | 1244.76 | **323.64** | 10 |

# 7  Determination of QS$_{sed}$ according to mesocosm/field data

No field or mesocosm studies that provide effect concentrations of AHTN in sediment are available, thus, no QS$_{sed}$ based on field data or mesocosm data has been derived.

# 8 Available sediment quality guidelines

As no sediment quality guidelines were identified, Table 8 instead summarizes PNEC values derived for freshwater sediments used in risk assessments.

The PNEC of 1720 µg/kg d.w. derived in EC (2008) is equal to the QS$_{sed,AF}$ derived here for 5 % OC, and in the same order of magnitude as the PNEC$_{EqP}$ derived for the EU risk assessment (EC 2008) and the PNEC$_{EqP}$ derived within the HERA (Human and Environmental Risk Assessment on ingredients of household cleaning products; HERA 2004) initiative, all ranging within a factor 5. The QS$_{sed,EqP}$ derived here for 5 % OC differ in one order of magnitude because the application of the additional AF of 10 to account for Log K$_{ow}$>5 is applied to exposure in HERA (2004). The differences with the PNEC from EC (2008) derived using the EqP is attributed to the K$_{oc}$ value and mainly to the different parameters in the PNEC$_{EqP}$ derivation used in EC (2008) compared to EU TGD (2018).

*Table 8 PNEC derived for freshwater sediments reported in the literature.*

| Description | Value [µg/kg d.w.] | Development method | References |
|---|---|---|---|
| PNEC | 5200 | Equilibrium partitioning, K$_{sed/wat}$ 746 m$^3$/m$^3$, based 36 d NOEC for *Pimephales promelas* (0.035 mg/L) and AF = 10. Additional factor of 10 to account for log K$_{ow}$>5 should be applied to exposure. Derived for sediments according to EU TGD EQS (EC 2018); sediment OC 5 % | HERA (2004) |
| PNEC | 1720 | AF method, NOEC growth of *Lumbriculus variegatus*, AF = 10; sediment OC 5 %. | EC (2008) |
| PNEC | 8420 | Equilibrium partitioning, based on 5 d EC$_{10}$ for *Acartia tonsa* (marine copepod) (0.026 mg/L) and AF = 10, derived for suspended solids according to EU TGD (EC 2003); sediment OC 5%. | EC (2008) |

# 9 Toxicity of degradation products

A number of (more polar) transformation products (TPs) are formed during biodegradation of AHTN, but that these TPs were not characterised (EC 2008). Detailed information on AHTN degradation products was not available for review, among others due to the original reports not being publically available. Available risk assessments do not cover environmental toxicology of degradation products. The EU risk assessment report only mentions the photosensitizing properties of "2 of 4" AHTN degradation products in Guinea pigs without details on the chemical nature being available (EC 2008).

Due to the lack of (a) characterisation of the TPs and (b) information on their ecotoxicity, it is conclusive that the present EQS$_{sed}$ proposal does not include a specific risk assessment for the TPs of AHTN.

# 10 EQS$_{sed}$ proposed to protect benthic species

The different QS values for each derivation method included in the EC EQS TGD 2018 are summarized in Table 9. According to the TGD, the most reliable extrapolation method for each substance should be used (EC 2018). In all cases, data from spiked sediment toxicity tests are preferred over the EqP approach.

Table 9 *QS_{sed} derived according to the three methodologies stipulated in the EU-TGD and their corresponding AF. All concentrations expressed as µg/kg d.w.*

|  | Sediment 1 % TOC | Sediment 5 % TOC | AF |
|---|---|---|---|
| QS_{sed,SSD} | - | - | - |
| QS_{sed,EqP} | 65 | 324 | 10 |
| QS_{sed,AF} | 344 | 1720 | 10 |
| EC EQS_{sed} | - | - | - |
| EC PNEC | - | 1720 | 10 |
| **Proposed EQS_{sed}** | **344** | 1720 | 10 |

A tentative EQS_{sed} of 344 µg/kg (1% OC) for AHTN including the application of an AF of 10 is thus proposed.

## 10.1  Uncertainty analysis

The EQS_{sed} derived here assumes that AHTN does not have ED effects. AHTN is scheduled for re-evaluation in the Rolling Action Plan by the EU member state Germany for being a potential endocrine disruptor and due to high (aggregated) tonnage (EC 2020). If the conclusion on ED protection changes, the EQS_{sed} may be reviewed to take this into account.

Due to the lack of characterisation of the transformation products of and information on their ecotoxicity, the present EQS_{sed} proposal does not include a specific risk assessment for the transformation products of AHTN.

According to current analytical capacities, LOQ is not limiting the application of the suggested EQS_{sed} in the uncertainty analysis

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## Appendix I. Sediment-water partition coefficient (K$_{oc}$) coefficient

| TOC, type | | Log K$_{oc}$ | K$_{oc}$ | Reference/Source |
|---|---|---|---|---|
| Field study, freshwater susp. matter; Elbe river, DE | exp. | 4.90 | 79433 | Winkler et al. (1998) cited in EC (2008) |
| Freshwater suspended matter | exp. | 4.40 | 25119 | Fooken (2004) cited in EC (2008) |
| Freshwater sediment | exp. | 4.00 | 10000 | Fooken (2004) cited in EC (2008) |
| Freshwater suspended matter, small streams with high input of waste water | exp. | 3.76 | 5754 | LfU-BW (2001) cited in EC (2008) |
| Freshwater suspended matter, small streams with high input of waste water | exp. | 4.65 | 44668 | LfU-BW (2001) cited in EC (2008) |
| Estimated from K$_{ow}$ (5.4) | est. | 4.47 | 29512 | log K$_{oc}$ = 0.81 x log K$_{ow}$ + 0.10 |
| | | **4.36** | **23085** | **Geomean** |