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Published in:
Environmental Science and Technology

DOI:
10.1021/es702500t

Citation for published version (APA):

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Chronic Exposure of the Oligochaete Lumbriculus variegatus to Polycyclic Aromatic Compounds (PACs): Bioavailability and Effects on Reproduction

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Received October 08, 2007. Revised manuscript received January 22, 2008. Accepted February 06, 2008.

This study aimed to monitor PAC availability to the oligochaete Lumbriculus variegatus during 28 days of exposure to spiked sediments, in order to obtain reliable chronic effect concentrations for reproduction. Sediment toxicity tests were performed using three pairs of PAC isomers: two homocyclic compounds (anthracene and phenanthrene), two azaarenes (acridine and phenanthridine), and the two main transformation products of the azaarenes (acridone and phenanthridone).

Introduction

Benthic invertebrates inhabiting PAC contaminated sediments are chronically exposed to a variety of homocyclic and heterocyclic compounds (1, 2). Risk assessment for PACs, however, is based on only a limited set of homocyclic compounds, ignoring the vast number of heterocyclic compounds (with in- or on-ring substitutions) and transformation products (3). Moreover, in the past PAC toxicity has been commonly assessed in short-term high-dose experiments, in which mortality is often the only end point (4). During such acute exposures PACs act mainly by narcosis (5), but during chronic exposure the same compounds may exert sublethal effects (on, e.g., growth, reproduction) (6, 7). Therefore, the present study focused on assessing chronic effects of azaarenes and their transformation products in addition to homocyclic compounds. Azaarenes are mainly formed and released into the environment by incomplete combustion of fossil fuels, in spills or effluents of several industrial activities, and by pesticide use (8). Azaarenes, containing one in-ring nitrogen substitution, can comprise up to 10% of the total PAC concentration of contaminated sites (9), but toxicity data for this group of compounds are scarce, and very little is known about their chronic effects (6, 7).

Sorption to sediment organic matter may decrease availability of PACs to benthic invertebrates, since tissue residues and effects correlate to dissolved pore water concentrations (10). Nevertheless, in chronic toxicity tests PAC bioavailability during exposure is rarely monitored, although changes in bioavailability could alter the outcome of the experiments (11). Recently, several techniques have been developed to facilitate quantification of the freely dissolved (pore) water concentrations of toxicants (12, 13), including solid-phase microextraction (SPME) (14). The poly(dimethylsiloxane) coating on the SPME fiber mimics the structure and sorptive properties of membranes (15), allowing a reliable estimation of the availability of organic compounds.

The aim of the present study was, therefore, to monitor PAC availability to the Oligochaete Lumbriculus variegatus during chronic exposure, in order to obtain reliable effect concentrations for reproduction. Asexual reproduction of L. variegatus is a sensitive chronic end point that can easily be combined with SPME and PAC accumulation measurements in the worms’ body in a single test. This allowed us to express the observed effects as a function of measured PAC concentrations in sediment, estimated PAC concentrations in pore water, and PAC concentrations in the organism. Since it has been demonstrated that slight differences in chemical structure may result in substantial differences in toxicity (16–18), three three-ring PAC isomer pairs were selected: two homocyclic compounds, two azaarenes, and the two main stable transformation products of the azaarenes.

Materials And Methods

Test Organism. Lumbriculus variegatus is an endobenthic oligochaete, widely used in bioaccumulation experiments (11, 19–21). In the laboratory it reproduces via fragmentation, and a standardized protocol to determine effects on asexual reproduction of L. variegatus has been developed by the Organisation for Economic Co-operation and Development (OECD). This protocol, with slight modifications, was applied in the present study.

Test Compounds. Six test compounds were chosen: two three-ring homocyclic compounds, anthracene and phenanthrene; two azaarenane analogues, acridine and phenanthridine, and two phase I azaarene transformation products, acridone and phenanthridone. Several properties of the compounds are listed in Table S1 in the Supporting Information. All compounds were provided by Sigma-Aldrich Chemie BV (Zwijndrecht, The Netherlands).

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10.1021/es702500t CCC: $40.75 © 2008 American Chemical Society Published on Web 03/22/2008
Sediment Spiking. Drontenmeer sediment, a Dutch reference sediment (12–14% organic matter, 15.5% clay, and all individual PAH concentrations below 0.01 mg/kg DW), was used for the toxicity tests. Sediment was provided by Gronenj/AquaSense (Amsterdam, The Netherlands). After sampling, it was homogenized and frozen to eliminate indigenous fauna (~20°C). Three days before the spiking procedure was started the sediment was thawed at 4°C.

The sediment was spiked to obtain the following nominal PAC concentrations: 67, 140, 280, 561, and 1122 µmol/kg DW for anthracene, 140, 280, 561, 1122, and 2244 µmol/kg DW for phenanthrene, 139, 279, 558, 1116, and 2232 µmol/kg DW for acridine and phenanthridine, and 154, 318, 640, 1281, and 2561 µmol/kg DW for acridone and phenanthridone.

Controls and solvent controls were included. The homocyclic PAC concentration series equal volumes of acetone were added to all treatments. The spiking was carried out in the dark to prevent photodegradation of the compounds. Controls and solvent controls were included. The homocyclic compounds and the azaarenes were dissolved in 50 mL of acetone and added to 38 g of dry sediment, and the mixture was left in a closed 500 mL glass bottle for 24 h to equilibrate. The transformation products, less soluble in acetone, were dissolved in 300 mL of acetone. After the 24 h equilibration period, the bottles were opened and left for 24 h in a fumehood to allow the acetone to evaporate. Next, the spiked dry sediment was mixed with 800 mL of Dutch Standard Water (DSW, deionized water with 200 mg/L CaCl2 and 100 mg/L MgSO4·H2O, 180 mg/L NaHCO3, 100 mg/L NH4Cl, and 20 mg/L KHCO3; pH 8.2) and 432 g of wet sediment (20% water content) in 3-L glass bottles. A 20:1 wt/wt mixture of Trouvit and TetraPhyll, approximately 10% water content, was placed on a roller bank (30 rpm) for 24 h to homogenize. Next, the mixture was divided into eight replicates 200 mL glass jars and allowed to settle for 10 days. Each test replicate consisted of approximately 60 g of wet sediment and 100 mL of DSW. Four replicates per concentration were sacrificed during the experiment for PAC measurements in sediment and pore water, leaving four replicates for the toxicity test.

Toxicity Tests. Four-week (28-day) toxicity tests were performed in an incubator at a constant temperature of 20 ± 1°C, with mercury lamps (light intensity approximately 50 µmol quanta/m²/s) providing a light regime of 16 h of light and 8 h of darkness. UV filters were used to minimize photodegradation of the test compounds by UV–B radiation. Glass jars were covered with plastic foil and constantly aerated with a gentle stream of nitrogen. All extracts were performed to determine values, as in ref (25).

PAC Concentrations in Sediment, Pore Water, and Oligochaetes. Sediment and pore water samples were collected at the beginning, after 7 and 14 days, and at the end of the experiment. For sediment samples, 40 mL was centrifuged for 15 min at 3000 rpm. To obtain a clear pore water sample, the supernatant was collected and centrifuged again for 15 min at 3000 rpm. Acetonitrile (150 µL) was added to the water sample, and the resulting mixture was vortexed for 30 s. Acetone was used as carrier solvent, and for each PAC concentration series equal volumes of acetone were added to all treatments. The spiking was carried out in the dark to prevent photodegradation of the compounds. Controls and solvent controls were included. The homocyclic compounds and the azaarenes were dissolved in 50 mL of acetone and added to 38 g of dry sediment, and the mixture was left in a closed 500 mL glass bottle for 24 h to equilibrate. The transformation products, less soluble in acetone, were dissolved in 300 mL of acetone. After the 24 h equilibration period, the bottles were opened and left for 24 h in a fumehood to allow the acetone to evaporate. Next, the spiked dry sediment was mixed with 800 mL of Dutch Standard Water (DSW, deionized water with 200 mg/L CaCl2·2H2O, 180 mg/L MgSO4·H2O, 100 mg/L NaHCO3, and 20 mg/L KHCO3; pH 8.2) and 432 g of wet sediment (20% water content) in 3-L glass bottles. A 20:1 wt/wt mixture of Trouvit and TetraPhyll, approximately 10% water content, was placed on a roller bank (30 rpm) for 24 h to homogenize. Next, the mixture was divided into eight replicates 200 mL glass jars and allowed to settle for 10 days. Each test replicate consisted of approximately 60 g of wet sediment and 100 mL of DSW. Four replicates per concentration were sacrificed during the experiment for PAC measurements in sediment and pore water, leaving four replicates for the toxicity test.

Toxicity Tests. Four-week (28-day) toxicity tests were performed in an incubator at a constant temperature of 20 ± 1°C, with mercury lamps (light intensity approximately 50 µmol quanta/m²/s) providing a light regime of 16 h of light and 8 h of darkness. UV filters were used to minimize photodegradation of the test compounds by UV–B radiation. Glass jars were covered with plastic foil and constantly aerated to keep oxygen concentrations stable. Once a week, deionized water was added to compensate for evaporation of the test solution. pH in the test jars ranged from 7.5 to 8.5 and oxygen concentrations were always above 75% air saturation.

Due to their low hydrophobicity, SPME could not be applied to the transformation products. The fibers, obtained from the Institute for Risk Assessment Sciences (IRAS, University of Utrecht, The Netherlands), had a diameter of 110 µm and a 28.5 µm thick poly(dimethyldisiloxane) (PDMS) coating, and were manufactured by Poly Micro Industries (Phoenix, AZ). Three-centimeter pieces of fiber were glued to a 0.4 mm Ø nylon fishing thread, washed with 70% methanol and deionized water and carefully introduced into the test sediment. Two replicate fibers were introduced into each test jar. Fiber measurements were performed in two jars per test concentration. The fibers remained in the sediment for three days, sufficient time for the fiber to reach equilibrium with the PCA concentrations in the pore water (23, 24). Next, they were gently cleaned with a tissue, and the fiber tip was cut to get rid of the glued tip of the fiber and extracted in 300 µL of acetonitrile. For each toxicity test, fibers were introduced into the sediment at the beginning and at days 3, 6, 9, 13, 17, 21, and 25 of the experiment. PAC concentrations in the fibers were measured by high-performance liquid chromatography (HPLC), using a 1 mL/min 80% acetonitrile/20% water flow rate and an injection volume of 20 µL. For the homocyclic compounds, fiber–water partition coefficients (Kd) from Ter Laak et al. (24) were used to estimate concentrations in the pore water using the SPME measurements. For the azaarenes, static system experiments were performed to determine values, as in ref (25).

PAC Availability. Availability of anthracene, phenanthrene, acridine, and phenanthridine in sediment pore water was measured using solid-phase microextraction (SPME).
rpm and the supernatant was transferred to a HPLC vial.
Acetonitrile (3 mL) was added to the pellet, the sample was
homogenized, and the extraction procedure was repeated.
The two worm tissue extracts were pooled and measured
using HPLC. PAC concentrations in the tissue were corrected
for recovery. Extraction recoveries were 70% for anthracene,
72% for phenanthrene, 65% for acridine, 67% for phenan-
thridine, 68% for acridone, and 72% for phenanthridone.

Statistical Analysis. The concentration of the test com-
ounds in sediment, pore water, and worm tissue that caused
50% reduction in reproduction compared to the control was
calculated according to Haanstra et al. (26). Average measured
PAC concentrations in the sediment are shown in Table S2. Concentrations of anthracene,
acridine, and acridone in the sediment did not decrease
substantially during the experiment (respectively, 3 ± 2, 3 ±
3, and 0%). In contrast, concentrations of phenanthrene,
phenanthridine, and phenanthridone in the sediment de-
creased during the experiment, mainly due to microbial
degradation (27, 28). Average decreases were 19 ± 10% for
phenanthrene, 30 ± 16% for phenanthridine, and 24 ± 12%
for phenanthridone.

PAC concentrations in pore water were estimated using
the concentrations in SPME fibers and PDMS-water partition-
ing coefficients (Kf, Table S1). In Figure 1, estimated
concentrations (Y-axis) were plotted against directly measured
PAC concentrations in pore water (X-axis), and a linear
regression was performed in order to evaluate the perfor-
mance of the SPME technique. The r² values of the linear
regression for anthracene, phenanthrene, and phenanthri-
dine were respectively 0.77, 0.92, and 0.97, showing a
consistent linear relationship between measured and esti-
mated PAC concentrations in pore water. For acridine, the
r² value was 0.64, due to variability in the results obtained
with SPME. If direct measurements in pore water would be
equivalent to concentrations in pore water estimated using
SPME, the slope of the regression line should be 1. This was
indeed the case for phenanthrene (1.07, 95% C.I. 0.89–1.19),
but for anthracene the slope was significantly higher than 1
(2.43, 95% C.I. 1.85–3.02), meaning that the measured
anthracene concentrations in the pore water were lower than
estimated ones. For the two azaarenes, acridine and phenan-
thridine, the slope of the regression line was significantly
lower than 1 (respectively 0.54 (95% C.I. 0.32–0.75) and 0.72
(95% C.I. 0.60–0.84)), meaning that estimated concentrations
were somewhat lower than concentrations measured in pore
water.

During exposure, estimated pore water concentrations of all PACs decreased at all concentrations in the sediment, due to increased sediment sorption in time and degradation by bacteria present in the sediment (28, 29) (Tables S3, S4,
In general, average PAC concentrations in the worms’ tissue (µmol PAC/kg wet weight) after 28 days of exposure increased with increasing PAC concentrations in the sediment, but differences in accumulation among the six tested compounds were found (Table S7).

**Effect Concentrations.** Control reproductive output after 28 days of exposure was on average 80% ± 9%. Control and solvent control average reproductive output did not differ significantly (t test, p < 0.05) and therefore solvent control values were used for further analysis of the results. All tested compounds except phenanthridone affected reproduction of *L. variegatus.* In Table 1, 50 and 10% effect concentrations (EC50 and EC10) with their 95% confidence intervals are shown. Likelihood ratio tests were performed to compare the EC50 values.

**EC50s Based on Measured PAC Concentrations in Sediment.** From the compounds that exerted an effect, acridone was the most toxic (EC50 301 µmol/kg sediment DW), while its parent compound acridine was the least toxic (EC50 1248 µmol/kg sediment). Likelihood ratio tests showed that the sediment EC50 for acridine was significantly higher than EC50s for phenanthridine, the homocyclic compound anthracene, and the transformation product acridone. In contrast, EC50s for anthracene, phenanthrene, phenanthridine, and acridone did not differ from each other.

**EC50s Based on PAC Concentrations in Pore Water.** Pore water EC50s were calculated using estimated PAC concentrations in the pore water for the homocyclic compounds and the azaarenes. For acridine, the pore water EC50 was estimated from the sediment EC50 using its sediment-water partitioning coefficient (Kow) as in Droge et al. (6). Pore water EC50s ranged from 0.148 µM for phenanthridine to 3.97 µM for acridine, and again likelihood ratio tests showed that the EC50 for acridine was significantly higher than the EC50s for the other compounds. EC50s for anthracene, phenanthrene, phenanthridine, and acridone did not differ significantly.

**EC50s Based on PAC Concentrations in the Organism.** Fifty percent effect body residues (ER50s) based on wet weight were calculated. Likelihood ratio tests showed that the ER50s for the two homocyclic compounds, anthracene and phenanthrene, did not differ significantly (116 and 88 µmol/kg WW, respectively). The ER50 for the azaarene acridine (137 µmol/kg WW) was similar to the homocyclic compound ER50s. In contrast, phenanthridine showed the significantly lowest ER50 (2 µmol/kg WW). The transformation product acridone showed a significantly lower ER50 than the homocyclic compounds and acridine (30 µmol/kg WW).

**Discussion.**

**PAC Availability to L. variegatus.** To our knowledge, this was the first attempt to use SPME to monitor the change in PAC availability in pore water during chronic toxicity tests. Indeed, the decrease in freely dissolved PAC concentrations in pore water during exposure time was substantial, according to previous observations on progressive sorption of PACs to particles (29, 30). Even more important, the present study showed that the freely dissolved PAC concentrations in pore water decreased more than total PAC concentrations in the sediment, emphasizing the importance of monitoring availability during chronic exposure.

For the two azaarenes, estimated concentrations using SPME were somewhat lower than directly measured concentrations in the pore water. This may have been caused by a slight underestimation of PDMS—water partitioning coefficients (Kow) due to depletion in the static experimental system in which Kow values were determined, as suggested by Poerschmann et al. (31). For anthracene, in contrast, concentrations measured directly in the pore water were 2.4 times lower than estimated pore water concentrations using SPME measurements. Due to the low water solubility of anthracene, binding of anthracene to particulate and dissolved organic carbon in the sediment pore water probably occurred during the pore water analysis (25). As a result, very low concentrations were measured. Therefore, pore water concentrations estimated using SPME measurements were more reliable, and were used to calculate effect concentrations.

**Effect Concentrations.** Effects Based on Measured PAC Concentrations in the Sediment (Figure 2A). Due to equilibrium partitioning, toxic effects of PACs will occur at lower concentrations in the (pore) water with increasing log Kow of the test compounds (32). On the other hand, sorption of the test compounds to sediment organic matter increases with increasing log Kow (24). Because of these two counteracting influences, sediment EC50s for our test compounds were expected to be more or less constant (33). This was indeed the case for the homocyclic compounds (anthracene and phenanthrene), the azaarenes (acridine and phenanthridine), and the transformation product acridone. The transformation product phenanthridone was the only compound that did not affect reproduction of *L. variegatus* at the tested concentrations (up to 4000 µmol/kg dry sediment). Sediment
EC50s were in the range of effect concentrations determined for other benthic invertebrates exposed to PAH-contaminated sediments (34–36).

**Effects Based on PAC Concentrations in the Pore Water (Figure 2B).** In previous studies (6, 35), estimated chronic pore water effect concentrations for soil and sediment invertebrates were compared to an acute LC50-log K_{ow} relationship determined by Bleeker et al. (5), which described the narcotic effects of several PACs to the midge *Chironomus riparius*. This way, effects of homo- and heterocyclic compounds were compared, and specific effects emerging besides narcosis during chronic exposure were identified. Applying this approach to the present data set revealed that pore water EC50s for the homocyclic compounds anthracene and phenanthrene agreed well with the acute LC50-log K_{ow} relationship. From the four tested heterocyclic compounds, however, only the EC50 for acridine was in agreement with the acute LC50-log K_{ow} relationship. The transformation product phenanthridone did not affect reproduction of *L. variegatus*. Based on its log K_{ow} value an EC50 of about 20 µM was expected, but this concentration in the pore water was not reached at the highest concentration in the sediment. In contrast, the EC50s for the azaarene phenanthridine and the transformation product acridone were clearly below the concentration corresponding to narcosis, suggesting a specific mode of action.

The specific effects found for the azaarene phenanthridine compared to the narcotic effects of its isomer acridine were observed previously for reproduction of the soil invertebrates *Folsomia candida* and *Enchytraeus crypticus* (6). Moreover, phenanthridine was found to be more teratogenic than acridine to *Xenopus laevis* (37), and more genotoxic in the GFP assay (38). Isoemer-specific toxicity was also found for the transformation product isomers: acridone was more toxic than expected based on log K_{ow}, as found previously in a mutagenicity study (18), while phenanthridone did not affect reproduction of *L. variegatus* at all at the highest concentration tested. In contrast, both transformation products affected emergence of *C. riparius* (35), underlining the species-specific sensitivity to PACs (6).

**Effects Based on PAC Concentrations in the Organism (Figure 2C).** Toxic effects can only take place if the toxicant is taken up by the organism. Therefore, we also expressed effects as a function of PAC concentrations in the organism. This allowed us to evaluate the observed deviations from narcosis in a second independent way, using the critical body residue (CBR) approach. The CBR concept (39) predicts that nonpolar narcotics (e.g., PACs, PCBs) cause 50% of mortality concentration.
at threshold body concentrations between 2 and 8 µmol/g wet weight. This concept has been successfully applied to interpret toxicity data for sediment- and soil-inhabiting organisms (4, 40–43). Because of the unselectivity of the narcotic mode of action, it has been suggested that this threshold would be applicable to all organisms. Furthermore, analyses of chronic toxicity data suggest that threshold body concentrations for chronic exposure would be 1 order of magnitude lower than lethal CBRRs (between 0.2 and 0.8 µmol/g wet weight) (34, 39, 43, 44), and body concentrations far below the threshold would indicate specific modes of action of the tested compounds occurring besides narcosis (45). In our study, ER50s for anthracene, phenanthrene, and acridine were around 1 order of magnitude lower than 2–8 µmol/g wet weight, in agreement with the chronic CBR concept. Also, ER50s for the two homocyclic compounds were close (within a factor 2) to body residue predictions for chronic exposure to homocyclic PAHs (43). In contrast, ER50s for phenanthidine and acridone were 2 orders of magnitude lower than the chronic CBR range, suggesting additional specific modes of action of the two compounds. Thus, deviations from narcosis found for the ER50s were in agreement with deviations from narcosis found for the pore water EC50s.

In this study, only one of the four tested heterocyclic PACs caused narcotic effects during chronic exposure, showing that (isomer) specific toxicity of azarenes and their transformation products could not be predicted from acute effect concentrations. About two-thirds of the approximately four formation products could not be predicted from acute effect caused narcotic effects during chronic exposure, showing agreement with deviations from narcosis found for the pore water EC50s.

Acknowledgments
This research was supported by the Technology Foundation (STW), applied science division of The Netherlands Organization for Scientific Research (NWO) (project AEB 06364).

Supporting Information Available
Tables S1–S7. This information is available free of charge via the Internet at http://pubs.acs.org.

Literature Cited


ES702500T