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# Population Consequences of Cadmium Toxicity in Soil Microarthropods

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Chronic toxicity experiments were performed using the collembolan *Orchesella cincta* (L.) and the oribatid mite *Platynothrus peltifer* (Koch), which were exposed to various levels of cadmium in the food (green algae). Experimental results were combined with life-history information to obtain realistic estimates of the intrinsic rate of population increase and its sensitivity to Cd stress. Chronic LC50 values for dietary exposure to Cd were estimated as 1.60 μmol/g for *O. cincta* and 3.18 μmol/g for *P. peltifer*. No-observed-effect concentrations (NOECs) for growth and reproduction were 0.042 μmol/g for *O. cincta* and 0.026 μmol/g for *P. peltifer*. The main effects of Cd were, however, different in each species. In *O. cincta*, Cd affected primarily female growth, without a direct effect on reproduction; in *P. peltifer*, the effect was primarily on reproduction. Uptake of Cd was higher in *P. peltifer* than in *O. cincta*, and caused a loss of zinc in the former species. As a consequence of their differing physiological responses to Cd, mites and collembolans also reacted differently in their population growth rates. The capacity for population increase of mites appeared to be rather sensitive to Cd, while collembolans were able, to some extent, to maintain their capacity for increase, in spite of toxic effects at the individual level. These results may contribute to developing ecotoxicological theory for the population consequences of toxic action in species with diverging life histories. Soil microarthropods may be suitable test organisms for evaluating the risk of chemicals to the soil ecosystem. © 1989 Academic Press, Inc.

## INTRODUCTION

A central theme of ecotoxicology concerns the relation between the toxic action of a chemical on individual organisms and the performance of a population of these organisms. The mechanisms that exist within a population to compensate or to amplify toxic effects on individuals are central to an understanding of environmental consequences of chemical pollution.

One of the approaches followed in population toxicity studies is the life-table or demographic method. From life-cycle toxicity experiments one estimates various population performance indices, such as the "intrinsic rate of population increase" and "carrying capacity" (Allan and Daniels, 1982; Gentile *et al.*, 1982; Bengtsson *et al.*, 1985; van Leeuwen *et al.*, 1987). Recently, a parameter called "biomass turnover ratio" has been proposed as another population performance statistic (van Straalen and de Goede, 1987). Rather than individual measures, population parameters offer a more realistic indication of the ecotoxicity of a chemical. Discussion has arisen around the question, whether these indices would be more sensitive to chemical stress

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than individual measures (growth, reproduction) or relatively insensitive. Some authors conclude that population parameters are generally less sensitive or at most equally sensitive, compared to individual parameters (Meyer *et al.*, 1987; Day and Kaushik, 1987). Others argue that some population parameters can be highly sensitive, because imperceptible effects on individuals are magnified by the great number of individuals constituting a population (Halbach *et al.*, 1983). This paper aims to demonstrate that this discussion is too limited; depending on the ecological context either argument may be true.

Responses of population parameters to toxic action of chemicals have been analyzed mostly for short-lived organisms with high fecundity (cladocerans, rotifers). Yet, from ecological theory (Stearns, 1976), it has become clear that population variables, such as the intrinsic rate of population increase, are not always influenced to the same degree by changes in mortality and fertility. Depending on the type of life history, population increase may, in some organisms, be influenced mostly by an increase in mortality and in other organisms by a decrease in fertility. This study, therefore, reports on toxicity experiments using two species (a collembolan and an oribatid mite) that live in the same habitat, but differ widely in their life histories. Detailed life-history information is used in combination with the experimental results to obtain realistic estimates of population parameters.

Collembola and Oribatida are microarthropods occurring abundantly in the surface layers of forest soils. The collembolan *Orchesella cincta* (L.) and the oribatid mite *Platynothrus peltifer* (Koch) are both common species from these groups. From ecological research (Weigmann, 1975; van Straalen, 1985a; Schenker, 1986) their life cycles can be summarized as follows. *O. cincta* (adult length 3 mm) has two generations per year, one recruited in spring and one in autumn. The species has separate sexes with indirect sperm transfer; females lay eggs at a rate of about 20 per week in laboratory cultures. Due to severe predation pressure in the field, the average lifetime is only a few weeks. *P. peltifer* (length 0.9 mm) has a single generation per year, recruited from July to November. The species is parthenogenetic. Diverging estimates are given for its fertility; in this study a maximal rate of three eggs per week was found. Adults may live for several months.

Within the diverse community of soil invertebrates, the two species can be seen as representing two separate points in a life-history continuum of short-lived, highly fertile species and long-lived, slowly reproducing species. Depending on their mode of action, persistent soil pollutants such as heavy metals will affect some life histories more than others and thus may change the community in a certain direction. Since many soil microarthropods are involved in the decomposition of soil organic matter (Seastedt, 1984), community changes may also alter the ecological functioning of the soil. By means of a comparative study, the present study contributes to developing ecotoxicological theory for the consequences of toxic action on species with diverging life histories.

## MATERIALS AND METHODS

### *Experimental*

Just-hatched Collembola [*Orchesella cincta* (L.)], about 3 days old, were taken from a laboratory culture and used directly for the experiment. The mites [*Platynothrus peltifer* (Koch)] were extracted by means of the Tullgren procedure from soil

cores taken in a pine stand of the Roggebotzand Forest, The Netherlands. The were kept for 1 week at 20°C before use in the experiment; only adults were used.

Collembola were reared in Perspex culture boxes (ϕ 5 cm) with a moistened plaster of Paris bottom, each box containing two randomly chosen individuals. The mites were kept in polythene culture boxes (ϕ 1 cm) with a bottom of chemically purified sand, covered with a piece of filter paper; each box contained 12 randomly chosen individuals. Both collembolans and mites were fed with green algae, which were prepared and contaminated as described by van Straalen *et al.* (1987); Cd was added to the food in a sulfate form. For each Cd addition, potassium sulfate was added, equalizing the sulfate concentration for all Cd doses. The food was freshly prepared each week.

In the experiment on Collembola, seven Cd treatments were applied, each being 30 times replicated (total: 420 individuals in 210 culture boxes). For the mites, six Cd treatments were applied, with eight replications (total: 576 individuals in 48 boxes). Dead animals were not replaced. The boxes were placed in a stratified random disposition in a climatized room at 20°C with a 12/12 light/dark cycle.

For *O. cincta*, the experiment lasted 61 days. Every ninth day the animals were weighed individually to the nearest microgram. During the reproductive period boxes were checked for eggs and hatchlings twice a week. For *P. peltifer*, the experiment lasted 84 days. Every seventh day the mites were transferred to a fresh culture box; the numbers of fecal pellets and eggs were counted, eggs being transferred to a new box and regularly inspected for hatchlings.

Individuals dying during the experiment plus those surviving at the end were frozen and stored for analysis. Metal analysis was done by graphite furnace atomic absorption spectrometry (Cd, Zn) and by flame atomic emission spectrometry (K). A microwave method for sample preparation was adapted from Bengtsson and Gunnarsson (1984). The collembolans were analyzed individually, while the mites were pooled in samples of four to six individuals. Food was analyzed at intervals during the experiments.

For statistical analysis, data from the collembolan experiment were sorted in male, female/female, and male/female pairs. Survivorship and molting were counted using all individuals. Growth was analyzed separately for the pairs of males and males of equal and unequal sex. Fertility (eggs per female) was averaged counting females from pairs of unequal sex. For mites, the culture box served as an experimental unit; fecal pellets and eggs were noted per box, following a correction for deaths.

LC50 values were estimated according to Kooijman (1981). The model fitted assumes a log-logistic distribution of the survival probability with dosage, and a hyperbolic change of the LC50 with time. NOECs were determined using the modified t test proposed by Williams (1971). Analysis of covariance (Sokal and Rohlf, 1981) was used to analyze the effect of Cd on clutch size, using female weight as a covariate.

### *Population Analysis*

Life-history data for *O. cincta* under uncontaminated field conditions were obtained from van Straalen (1985a,b). For each of the two successive generations in a year (spring and autumn generations), survival and fertility varied with developmental stage (weight) but were assumed invariant with time. For *P. peltifer*, life-history data were estimated from stage-frequency observations in Weigmann (1975). Stage-specific mortality and stage durations were assumed time-invariant, while fertility

was assumed to vary each month, with a peak in June, July and zero fertility from November through March. Based on Schenker (1986), it was assumed that adults live 3 months. Seasonal stage-structure changes predicted on the basis of these assumptions were in reasonable agreement with the data from Weigmann (1975).

Observations from the experiment and field data were combined by calculating a "stressed life cycle" for each Cd exposure by means of the following formulas:

$$\hat{l}_{ij} = \frac{\bar{l}_i}{l_{i0}} l_{ij} \quad \text{and} \quad \hat{m}_{ij} = \frac{\bar{m}_i}{m_{i0}} m_{ij},$$

where

$\hat{l}_{ij}$ = survival to stage (weight class) $i$ at Cd exposure level $j$ in the "stressed life cycle,"
$\bar{l}_i$ = survival to stage $i$ in the "unstressed" field life cycle,
$l_{i0}$ = survival to stage $i$ at zero Cd exposure (control) in the toxicity experiment,
$l_{ij}$ = survival to stage $i$ at Cd exposure level $j$ in the toxicity experiment,

and $\hat{m}_{ij}$, $\bar{m}_i$, $m_{i0}$, and $m_{ij}$ are the corresponding fertility parameters.

Thus, the experimental control situation is equal to the uncontaminated field situation, each effect of Cd on survival and fertility being accounted for in the life cycle by the same percentage as observed in the experiment. Since for *P. peltifer* only adults and eggs were used in the experiment, effects of Cd on survival of nymphal stages were assumed equal to those observed in the adults.

From the "stressed life cycles" the intrinsic rates of population increase over 1 year were calculated for *O. cincta* as $r = \ln(R_{01}R_{02})$, where $R_{01}$ and $R_{02}$ are the net reproduction rates ($R_0$) for spring and autumn generations, respectively, calculated as

$$R_0 = \sum_i l_i m_i.$$

For *P. peltifer*, the assumed time-varying fertility pattern necessitated a somewhat different approach. For each month, the development of stage structure was described by means of a projection matrix identical to the one considered by Longstaff (1977). Then a single year-to-year projection matrix was derived from the monthly matrices by matrix multiplication. From the single matrix, the yearly rate of population increase was calculated as $r = \ln \lambda$, where $\lambda$ is the leading eigenvalue of the year-to-year projection matrix. For sake of brevity, details of the matrix model are not given here but can be obtained from the authors on request.

## RESULTS

### *Uptake of Cadmium*

When cadmium was added to the food, the actual concentrations (measured on various samples during the experiments) were often lower than the nominal concentrations (Table 1). This may be due to the water-soluble fraction leaching from the food into the substrate. However, the controls contained Cd at a background level of about 0.01 $\mu$mol/g, even when the algae (*Pleurococcus*) were collected from a remote rural field site. Addition of Cd did not change the concentration of zinc in the food and addition of $K_2SO_4$ (added to equalize the sulfate concentrations of all dosages) failed to cause a significant increase in the potassium concentration (Table 1). The

TABLE 1

NOMINAL AND ACTUAL CONCENTRATIONS OF Cd, Zn, AND K IN FOOD (GREEN ALGAE) DOSED WITH $CdSO_4$ AND $K_2SO_4$

| Nominal Cd (μg/g) | Nominal Cd (μmol/g) | Actual Cd (μmol/g) Expt. *O. cincta* | Actual Cd (μmol/g) Expt. *P. peltifer* | Actual Zn (μmol/g) | Actual K (μmol/g) |
|---|---|---|---|---|---|
| 0 | 0 | 0.009 | 0.012 | 0.93 | 62 |
| 0.5 | 0.004 | 0.011 | — | — | 67 |
| 2 | 0.018 | 0.020 | 0.026 | 1.00 | 68 |
| 8 | 0.071 | 0.042 | 0.077 | 1.08 | 64 |
| 32 | 0.285 | 0.126 | 0.243 | 1.03 | 82 |
| 128 | 1.14 | 0.499 | 1.04 | 0.99 | 73 |
| 512 | 4.56 | 1.84 | 3.89 | 0.98 | 57 |

exchangeable fractions of elements in the contaminated food were 15–40% for Cd, 10–25% for Zn, and 100% for K. These are normal values for contaminated leaf litter in the field (van Straalen *et al.*, 1987). It may be concluded that chemical binding and the complexation of cadmium in laboratory contaminated food is not very different from that of the field. In the following, all effects are related to actual total concentrations on the basis of dry weight.

For *Orchesella cincta*, Cd levels in males and females did not differ significantly and were averaged. Both *O. cincta* and *Platynothrus peltifer* took up Cd in a concentration-dependent manner (Fig. 1). Since the uptake phase takes 3 to 4 weeks for both species (M.P.M. Janssen, unpublished), the concentrations reported here (9 and 12 weeks exposure time) probably represent equilibrium values. *O. cincta* appears to maintain a level below the food concentration, while *P. peltifer* concentrates the cadmium, relative to the food levels (cf. Fig. 1). This is consistent with field observations.

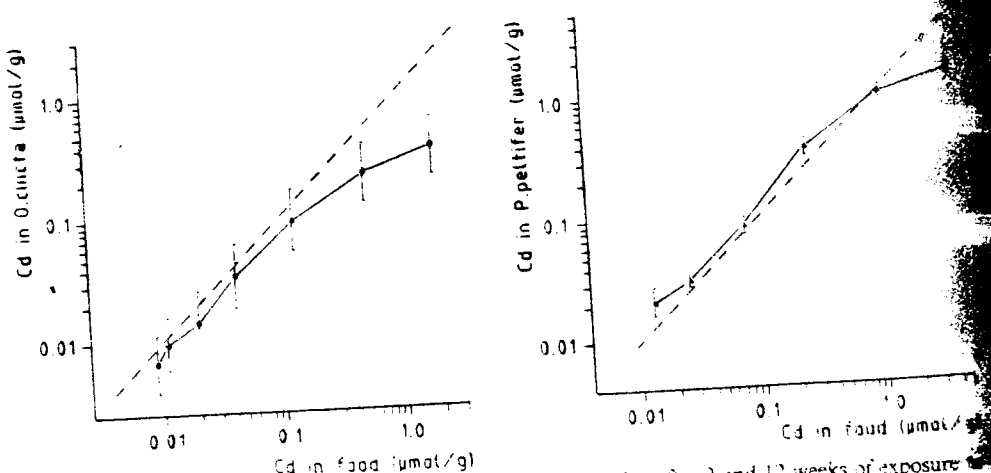


FIG. 1. Cd concentrations of *O. cincta* (left) and *P. peltifer* (right) after 9 and 12 weeks of exposure to Cd, respectively. Means are given with their standard errors ($n$ = 20 for *O. cincta*, males and females combined; $n$ = 8 for *P. peltifer*). The broken line indicates equal concentrations for animals and food.

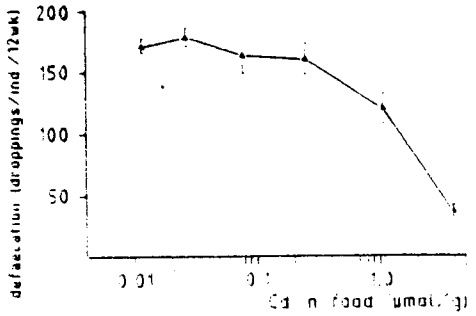


FIG. 2. Defecation of *P. peltifer* during the experiment as a function of the Cd dosage. Means are given with their standard errors, based on eight replicate boxes.

on the same species (Janssen, 1988). At the highest concentrations, both species take up less Cd than expected in proportion to the food concentration. As there are no indications of a concentration dependency of Cd excretion (van Straalen *et al.*, 1987), the lower levels attained here probably result from decreased consumption. In *P. peltifer*, direct evidence for this was obtained by measuring defecation (Fig. 2). Furthermore, in choice experiments (not reported here) *P. peltifer* was able to discriminate between clean food and food containing Cd at 3.9 µmol/g. Decreased consumption and behavioral avoidance of contaminated food has also been found for other metals and other species (Joosse and Verhoef, 1987; van Capelleveen *et al.*, 1986). It thus seems a common reaction among soil invertebrates.

The zinc contents of *P. peltifer*, exposed to Cd, appeared to be lower than those of control animals. Some Zn seems to be exchanged for Cd and is lost from the animal. The correlation between Zn and Cd in individual samples of mites was rather scattered (Fig. 3). At high Cd loadings (>1 µmol/g), no equimolar exchange for Zn seems to take place. It can be concluded that Cd exposure disturbs the Zn metabolism of *P.*

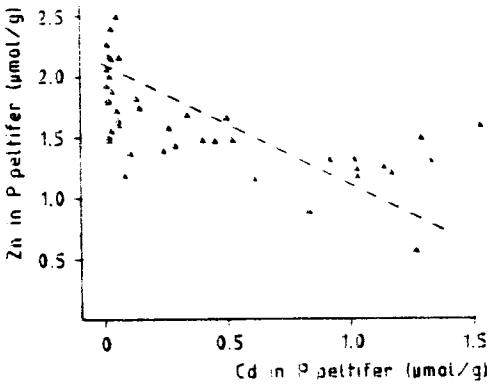


FIG. 3. Correlation between Zn and Cd in *P. peltifer* after 12 weeks exposure to various Cd dosages. Each point is a single sample, consisting of six pooled individuals. The broken line indicates an equimolar exchange of Zn for Cd at a constant Zn + Cd content (2.1 µmol/g, being the average Zn + Cd content of unexposed mites).

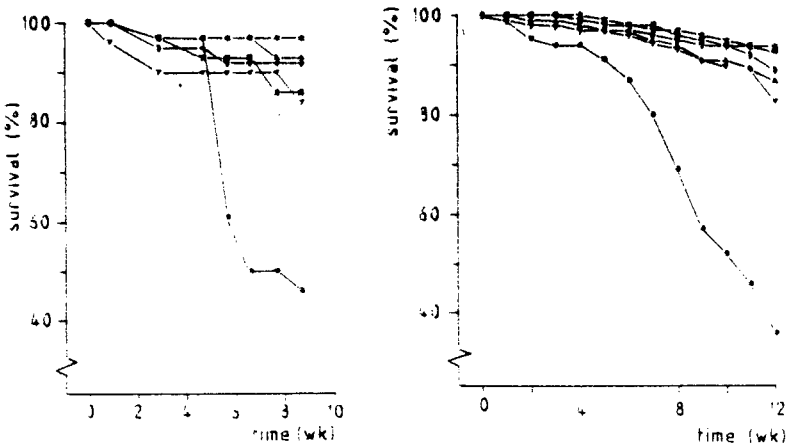


FIG. 4. Survival of *O. cincta* (left) and *P. peltifer* (right) at various Cd concentrations. ●, control; ■, 2 μg/g; ♦, 8 μg/g; ▲, 32 μg/g; ▼, 128 μg/g; ●, 512 μg/g (nominal concentrations). For actual concentrations, Table 1.

*peltifer*. In *O. cincta*, the Zn content was lower (on the average 1.27 μmol/g) than *P. peltifer*; no interference of Cd with Zn has been observed in this species.

*Mortality*

Cadmium caused significant mortality only in the highest exposure groups (Fig. 4). In *O. cincta*, mortality commenced abruptly after 5 weeks (Fig. 4, left), while in *P. peltifer*, mortality increased gradually (Fig. 4, right). In *O. cincta*, no difference between males and females was observed. Chronic LC50 values were estimated from these data and are given in Table 2. *P. peltifer* seems to be more tolerant to Cd than *O. cincta*, although the statistical significance of this difference is uncertain in the light of the wide confidence interval.

TABLE 2

ESTIMATES OF MEDIAN LETHAL CONCENTRATIONS OF Cd FOR *Orchesella cincta* AND *Platynothrus peltifer* AFTER 9 AND 12 WEEKS OF DIETARY EXPOSURE

| | *O. cincta* | *P. peltifer* |
|---|---|---|
| 9-Week LC 50 (μmol/g) | 1.60* | 7.27<br>3.79–13.96 |
| 12-Week LC50 (μmol/g) | — | 3.13<br>2.17–4.65 |
| Slope parameter | 0.42<br>0.26–0.56 | 1.10<br>0.81–1.38 |

*Note.* LC50's and slopes were estimated assuming the model in Kooijman (1981). Below each estimate, a 95% confidence interval is given, except for the value marked by an asterisk, where this could not be calculated due to model inadequacy.

TABLE 3

CONCENTRATIONS OF Cd IN ANIMALS SURVIVING THE EXPERIMENT AT THE HIGHEST Cd DOSAGE AND IN THOSE DYING DURING THE EXPERIMENT AT THE SAME DOSAGE

| | Survivors (μmol/g) | | | Dead (μmol/g) | | |
|---|---|---|---|---|---|---|
| | Mean | SE | No. observations | Mean | SE | No. observations |
| *Orcheseila cincta* | 0.36 | 0.027 | 20 | 0.55 | 0.053 | 11 |
| *Platynothrus peltifer* | 1.36 | 0.060 | 4 | 2.24 | 0.221 | 8 |

Cadmium levels of animals dying during the experiments were significantly higher than the levels of those surviving (Table 3). The concentrations in *P. peltifer* are higher than in *O. cincta*, both alive and dead. The two species clearly die from Cd intoxication at different internal concentrations, *P. peltifer* tolerating a larger internal dose than *O. cincta*.

### *Growth and Development*

Growth data for male/female pairs of *O. cincta* are presented in Fig. 5; only surviving pairs are included. Weight increase was clearly affected by Cd in the females, but not in the males (Fig. 5). The retarding effect on female growth appeared at about 4

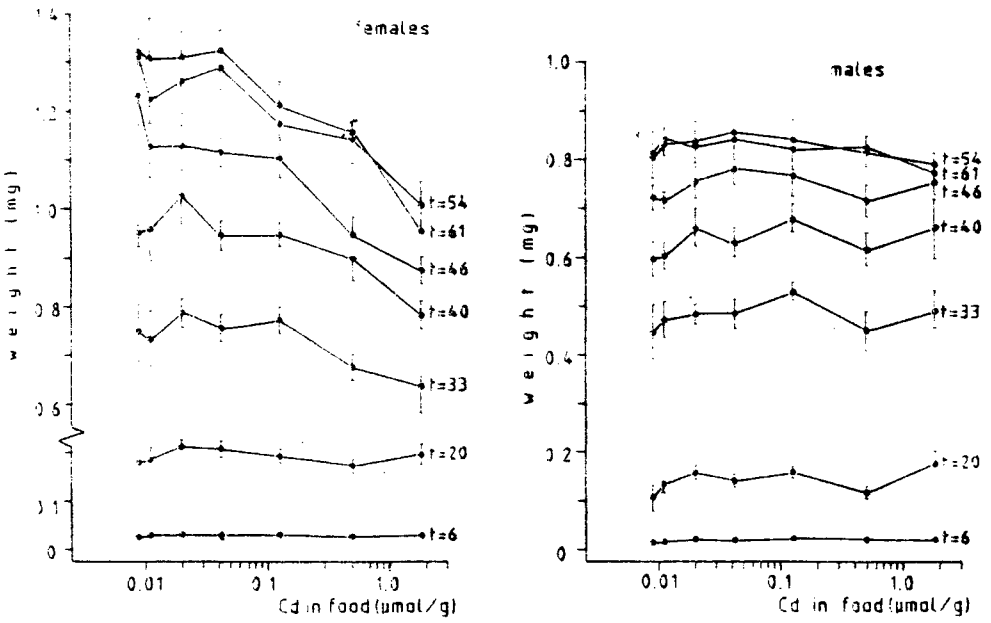


FIG. 5. Growth of female (left) and male (right) *O. cincta* at various times ($t$ = 6 days to $t$ = 61 days) during the experiment, as a function of Cd exposure. Means and standard errors are given based only on individuals surviving the complete experiment.

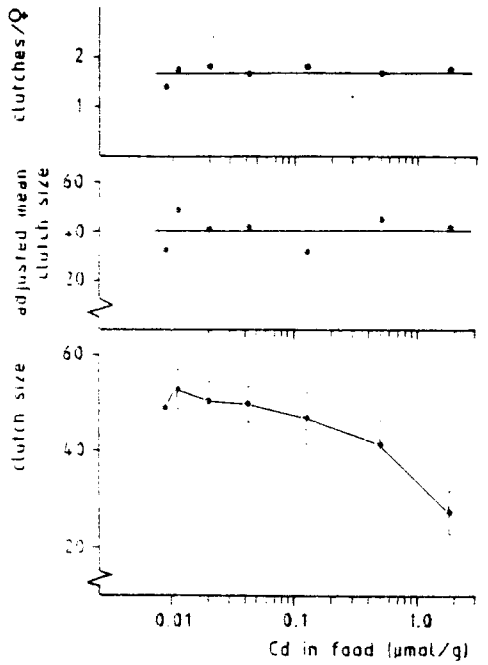


FIG. 6. Reproductive parameters of *O. cincta* at the various Cd concentrations. Mean clutch sizes with standard errors are based on all clutches laid per treatment. Adjusted mean clutch sizes are calculated following analysis of covariance, assuming a linear relationship between clutch size and weight. The average number of clutches per female is based on all male/female pairs, including those not reproducing.

weeks and continued to increase (Fig. 5, left). On the basis of the eventual weight attained by surviving females, the highest exposure concentration causing no effect was determined to be 0.042 µmol/g. The growth-inhibiting effect of Cd was also observed in female/female pairs of *O. cincta*, which of course laid no eggs. Therefore, it is not the female activity (egg production) that makes females sensitive to Cd, but the being female itself.

The average molting rate of male/female pairs of *O. cincta* was 1.23 per animal per week; no effect of Cd on molting was observed, either in males or in females. Weight increase in *P. peltifer* was minimal and not related to Cd exposure. Since adults have a fixed size and do not molt, weight increase is not a good parameter for this species.

### *Reproduction*

Eggs produced by male/female pairs of *O. cincta* were laid in clutches of variable size; the average clutch size of control animals was 50, which is in agreement with Janssen and Joosse (1986). The size of clutches gradually decreased with increasing Cd exposure, but only at the highest concentration was this statistically significant (Fig. 6: NOEC = 0.50 µmol/g). Since clutch size of *O. cincta* is dependent on the weight of the female, especially for the first clutch (Janssen and Joosse, 1986), a correction for female weight was applied by means of analysis of covariance. The weight-adjusted clutch sizes did not then show a trend for Cd exposure (Fig. 6).

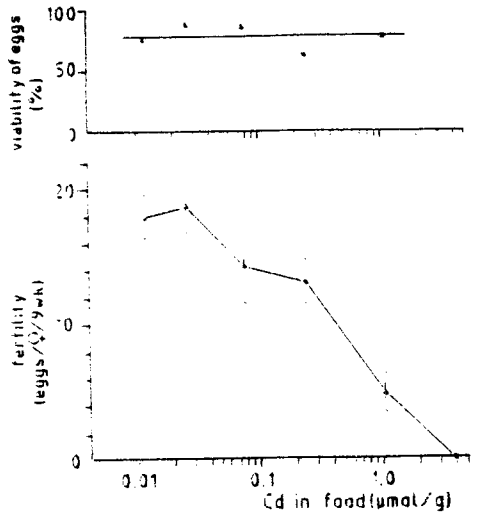


FIG. 7. Reproductive parameters of *P. peltifer* at the various Cd concentrations. Means with standard errors are based on the total number of eggs layed in each of eight boxes per treatment.

The number of clutches per female (counting females from all male/female pairs) was lowest for the control group, being due to the fact that reproduction started later in this group. Low-dose Cd exposure seems to have an accelerating effect on reproductive development; otherwise, Cd has no effect on clutch numbers (Fig. 6). Likewise, no effects of Cd on development time for eggs (±6 days) or on viability (±90%) were noted. It can be concluded that Cd does not affect reproduction of *O. cincta* directly; only through its effects on growth, is reproduction reduced.

*P. peltifer* commenced producing eggs in the fourth week, continuing to do so for the rest of the experiment. Since each box contained 12 parthenogenetic individuals, clutches laid by individual animals could not be discerned. Instead, the total number of eggs laid in a box was, after correction for the number of animals alive, cumulated over 9 weeks. Cadmium had a drastic effect on the reproduction of *P. peltifer*. At the highest exposure group, no eggs at all were laid (Fig. 7). The NOEC is 0.026 µmol/g.

The developmental times of eggs to larvae were extremely variable (20 to 60 days). Viability of eggs also varied between groups but not in a consistent pattern related to Cd (Fig. 7). It is concluded that Cd has an inhibiting effect on the reproduction of *P. peltifer*, but it does not affect the eggs after they are laid.

### *Population Growth*

Demographic data of *O. cincta* and *P. peltifer* under uncontaminated field conditions are summarized in Tables 4 and 5. The life history of the collembolan *O. cincta* (Table 4) is characterized by a very high rate of mortality among hatchlings, followed by a phase of lower, steady mortality; approximately 4% of the spring generation and about 2% of the autumn generation reach sexual maturity. Fertility is rather high. The life history of the oribatid mite *P. peltifer* is characterized by a low mortality (30% reach sexual maturity), but the fertility is also low (Table 5). Net reproduction rates ($R_0$), estimated from these tables, are $R_0 = 1.40 \times 1.30 = 1.82$ for *O. cincta* and

TABLE 4

LIFE TABLE FOR *Orchesella cincta* UNDER UNCONTAMINATED FIELD CONDITIONS

| Stage | Fresh weight (mg) | Stage duration (weeks) | Spring generation $l_i$ | Spring generation $m_i$ | Autumn generation $l_i$ |
|---|---|---|---|---|---|
| 1 | <0.02 | 1 | 1 | 0 | 1 |
| 2 | 0.02–0.06 | 2 | 0.299 | 0 | 0.149 |
| 3 | 0.06–0.1 | 2 | 0.237 | 0 | 0.0481 |
| 4 | 0.1–0.2 | 2 | 0.186 | 0 | 0.0358 |
| 5 | 0.2–0.5 | 3 | 0.120 | 0 | 0.0236 |
| 6 | 0.5–0.7 | 4 | 0.0388 | 33 | 0.0185 |
| 7 | 0.7–0.9 | 6 | 0.0027 | 46 | 0.0131 |
| 8 | >0.9 | 6 | 0 | — | 0.0018 |

*Note.* $l_i$ = survivorship to class $i$. $m_i$ = number of viable eggs produced per adult in class $i$. Data were taken from van Straalen (1985a, b).

$R_0$ = 1.46 for *P. peltifer;* from these values the population growth rate ($r$) is estimated as $r$ = 0.60 per year for *O. cincta* and $r$ = 0.38 per year for *P. peltifer*. For average field populations, $R_0$ will equal one and $r$ will be zero; the values given here were estimated under conditions where populations showed a small increase.

The effects of Cd on survival and reproduction were analyzed by superimposing them on the above field data (see Materials and Methods). For *P. peltifer*, fertility was not fixed at the value shown in Table 5, but varied between 0 and 3.5, according to season. Since the experiment on *P. peltifer* did not cover larvae and nymphs, it was assumed that Cd had an equal effect on mortality in all stages. In this manner for both species, a "stressed life cycle" for each Cd concentration was obtained from which population growth rates were calculated.

TABLE 5

LIFE TABLE FOR *Platynothrus peltifer* UNDER UNCONTAMINATED FIELD CONDITIONS

| Class | Stage | Stage duration (months) | $l_i$ | $m_i$ |
|---|---|---|---|---|
| 1 | Egg | 1.5 | 1 | 0 |
| 2 | Larva | 2 | 0.93 | 0 |
| 3 | Protonymph | 1.5 | 0.59 | 0 |
| 4 | Deutonymph | 4 | 0.51 | 0 |
| 5 | Tritonymph | 2 | 0.41 | 0 |
| 6 | Adult | 1 | 0.30 | 2.0 |
| 7 | Adult | 1 | 0.24 | 2.0 |
| 8 | Adult | 1 | 0.19 | 2.0 |

*Note.* $l_i$ = survivorship to class $i$. $m_i$ = number of eggs produced per adult in class $i$. Data were estimated from Weigmann (1975) and Schenker (1986).

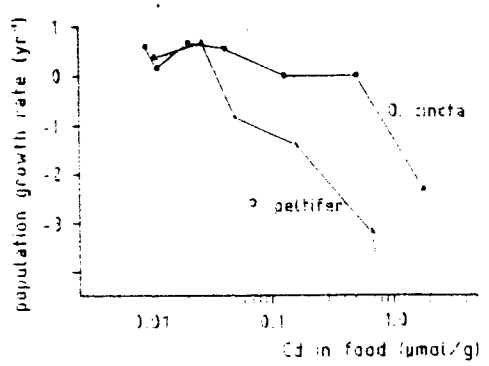


FIG. 8. Population growth rate of *O. cincta* and *P. peltifer*, calculated from toxicity data combined with life-history data from the field, as a function of Cd stress.

Figure 8 shows that the population growth rate of *P. peltifer* is more affected by Cd than the population growth rate of *O. cincta*. In fact, the latter species can maintain a positive $r$ up to a concentration of 0.5 µmol/g, while *P. peltifer* shows a positive $r$ only up to 0.03 µmol/g.

The NOEC values for all parameters studied are summarized in Table 6. It appears that the most sensitive of individual criteria (growth for *O. cincta*, reproduction for *P. peltifer*) have NOEC values in the same order of magnitude. However, the two species differ widely when judged by the population growth rate. With increasing Cd stress, *O. cincta* die before *P. peltifer*, but *P. peltifer* are affected before *O. cincta* with regard to population growth.

## DISCUSSION

This study has demonstrated that population performance indices such as the intrinsic rate of population increase are both sensitive and insensitive to toxic action on the individual level. Although the two species considered appear equally sensitive to cadmium if judged by the most sensitive of individual criteria (growth, reproduction), they differ in sensitivity when judged by the population growth rate. *P. peltifer*

TABLE 6

NO-OBSERVED-EFFECT CONCENTRATIONS OF Cd FOR *Orcheseila cincta* AND *Platynothrus peltifer* UNDER CHRONIC DIETARY EXPOSURE

| | NOEC (µmol/g) | |
|---|---|---|
| Criterion | *O. cincta* | *P. peltifer* |
| Survival | 0.50 | 1.04 |
| Consumption | 0.50 | 0.24 |
| Growth | 0.042 | — |
| Reproduction | 0.50 | 0.026 |
| Population growth | 0.50 | 0.026 |

has a higher LC50 than *O. cincta*, but a lower NOEC for population growth, dem
strating that a simple relationship between toxicant tolerance and life-history a
butes probably does not exist. A similar conclusion may be drawn from the ecotox
logical literature.

Neuhold (1987) analyzed LC50 data on fish in relation to longevity and fecundi
Although it was concluded that fertile and short-lived species tend to be more toler
than less fertile and long-lived species, there was also a great amount of unexplain
variation in the data. Meyer *et al.* (1987) pointed out that in a toxicological sense, t
sensitivity of a parameter is determined not only by its magnitude of change in re
sponse to the toxicant, but also by the statistical variability of its estimate. In clado
eran toxicity tests using Cu and Cd, none of the variables considered was consistent
the most sensitive (Meyer *et al.*, 1987). It is clear that the question of sensitivity vers
insensitivity of population parameters cannot be answered in a general sense. Popul
tion indices calculated from toxicity studies may not be directly representative of u
field situation, since regulating factors (food, predators) are not taken into accou
Kooijman (1985) argued that in populations regulated by predators, popula
growth will be most sensitive to toxicants affecting reproduction, while in populatio
regulated by their food, population growth will be most sensitive to effects on surviv

From the above conclusions, the results of this study demonstrated that not
effects on the individual level are of equal weight: *P. peltifer* populations demonstr
sensitivity to Cd because the primary effect is on reproduction, while in *O. cin*
populations the response is relatively insensitive because Cd does not directly af
reproduction. Bengtsson *et al.* (1983, 1985) concluded that population growth of t
collembolan *Onychiurus armatus*, stressed by lead and copper, was affected prima
by the effects of the metals on egg laying. Reproduction seems to be the most ecolog
cally relevant criterion for judging the effects of toxicants on soil microarthropods.

Various authors have reported changes in the soil microarthropod community i
gradient studies of metal-contaminated forest soils. Mites decrease with increasin
site contamination, while springtails often are not affected (Williams *et al.*, 197
Strojan, 1978; M. P. M. Janssen, unpublished). The present results may provide
explanation for these observations: Collembola such as *O. cincta* may survive ca
mium stress as a population, in spite of toxic effects on individual growth; mites a
affected by their sensitive population response. It must, however, be added that, bo
within Collembola and Oribatida, significant differences between species can be ex
pected. Collembola such as *O. cincta*, which inhabit the superficial layers of the fore
floor, seem to contain lower levels of metals than species such as *O. armatus*, whic
live deeper in the soil profile (cf. Bengtsson *et al.* 1983; Janssen, 1988). Among ori
batid mites, species variation in sensitivity to copper was demonstrated by Sta
(1984). *P. peltifer* populations decreased in soils containing 200 μg/g Cu, while s
other species did not.

Different physiological mechanisms may underlie the different effects of Cd on
mites and collembolans. In *P. peltifer*, Cd disturbed Zn metabolism (Fig. 3). Since
Zn is often related to reproductive functions (Williams, 1984), this may explain the
reproduction-inhibiting effect of Cd. In *O. cincta*, Cd inhibits the growth of female
without affecting egg laying directly (Figs. 5, 6). In this species, Cd toxicity must be
based on an entirely different mechanism, for example, a disturbance of hormone
promoting female growth. Collembolans are able to maintain low body concentra

tions of metals by means of excretion through intestinal exfoliation (Joosse and Buker, 1979; van Straalen *et al.*, 1987). No such mechanisms are known for mites.

In comparison to the well-developed field of aquatic toxicology, ecotoxicology of soil organisms is only just beginning. Most toxicity data are known for earthworms; only little experimental information is available for soil arthropods (isopods, mites, springtails). Yet, this group includes several easily bred and manageable species that may be suitable test organisms. Considering the great diversity of the soil community plus the variety of ecological niches occupied, the hazards of soil contaminants should be judged on toxicity data from a representative set of species. The present results may contribute to a strengthening of the basis of a proper risk evaluation for soil contaminants.

## CONCLUSIONS

Comparative toxicity research on species with diverging life histories may contribute to developing ecotoxicological theory for population responses to chemical exposure. For soil microarthropods, effects on reproduction seem to be the most relevant criterion. Soil arthropods may be useful test species for risk evaluation of soil contaminants.

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