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Costs of Living with Contaminants: Implications for Assessing Low-Level Exposures

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INTRODUCTION

Chemical toxicants challenge biological systems. A *priori* it is likely that responses to contaminants are costly for the organism in terms of metabolic resources and especially energy (Calow 1989). Here, we review theoretical and empirical evidence for this cost hypothesis, and then consider the implications for individual organisms, for evolutionary processes, and for understanding possible linkages between the responses of individual organisms to chemical exposure and responses occurring at ecological levels of organization.

The responses of organisms to high contaminant exposure concentrations are fairly obvious (e.g., gross morphological or physiological changes or death). Responses to low level exposure may be less easily discernible, involving more subtle responses, such as the induction of catabolic enzymes and/or protective proteins. However,

according to the cost hypothesis these responses are likely to have consequences for the individual's energy budget that, especially in summation over the life cycle, could be ecologically important. So here we consider the implications of the cost hypothesis for the ecological relevance of responses to low-level exposure.

In what follows we first classify the various types of energy demanding responses to chemical toxicants before we present critical reviews of the evidence for costs associated with exposure to them. We then discuss the likely implications of these costs for population and then community/ecosystem levels of organization. Finally we address the extent to which costs are helpful in elucidating the ecological relevance of responses to low-level exposure.

TYPES OF ENERGY-DEMANDING RESPONSES

Organisms encountering a toxic substance in the environment possess different strategies for dealing with the chemical, all of which are likely to demand some energy expenditure. The strategies are of two general types: inducible or constitutive. Inducible strategies involve any changes in behavior, physiology or biochemistry that only occur after exposure has been initiated (often following a measurable time-lag). Constitutive strategies involve morphological, physiological or biochemical features that persist irrespective of exposure. The costs associated with either of these strategies may be short term or long term. Inducible strategies will usually involve short-term costs, but they may be long term, for example under conditions of chronic exposure. Constitutive strategies will involve short-term constructional costs and possibly long-term maintenance costs.

The strategies used by organisms to resist chemical exposure can also be grouped into several other general categories: avoidance or escape reactions; exclusion (for example, many aquatic animals exposed to toxic chemicals secrete mucus onto exposed surfaces); removal (in-coming toxicants might be actively pumped out); detoxification, possibly followed by excretion (e.g., by sequestration in granules or via metabolic transformation); and repair of damage caused by toxicants. If these responses fail, there will be irreversible damage and pathological effects, leading ultimately to death. Selye (1950, 1973), Hatch (1962) and several others (Hellawell, 1986) have proposed a graded series of these responses to increasing concentrations of, or times of exposure to, toxicants, i.e., from avoidance, through exclusion, removal, detoxification, to repair as the intensity of exposure increases. To date, empirical data to support graded response models are rare, and we still have a poor understanding of the shape of the response curves with respect to increasing concentration or time of exposure.

EVIDENCE
From specific responses

Cost of Moving Away

Escape responses are inducible and hence are likely to involve largely short-term costs. The relationships between activity and metabolic rate are well established particularly in vertebrates and flying insects (Brafield and Llewellyn 1982). The net cost of transport (J kg^{-1} m^{-1}) is a decreasing function of body mass; for a given body mass it appears that the cost of movement increases in the following order: swimming <flying < running < burrowing (Alexander 1975; Trevor 1978). Thus attempts to move out of a contaminated area would be least costly for fish but particularly costly for burrowing organisms, and since most burrowers are relatively small, they would have to move a greater number of body lengths to enjoy the benefits of their increased activity (i.e., to effectively escape from a contaminated area).

Behavioral responses can be sensitive to very low-level chemical exposures, and marine species in particular appear able to detect low concentrations of novel substances (Kittredge 1980). Although more mobile species, such as fish, may respond to chemical exposure by moving out of the contaminated area, other species may not be capable of movement over relevant spatial scales or may respond to chemical exposure by reducing or ceasing movement. Reducing activity (and the fitness costs that go with it) can be an effective strategy if exposure is of a temporary nature and may make energy available for other stress resisting processes (e.g., enzyme production). Over longer-term exposures lowered activity is likely to indicate impaired performance e.g., in the form of reduced feeding and mating activity.

Cost of Keeping it Out

These may involve the induction of protective secretions such as mucus and/or morphological changes such as thickened body surfaces. The production of mucus may inhibit uptake of toxic chemicals but may entail a substantial cost. Several studies have estimated the cost of mucous production, under normal conditions, in molluscs and these have ranged from 20% of ingested energy in two species of freshwater snails (Calow 1974), to 70% of assimilated energy in *Chiton pelliserpentis* (Horn 1986) to 80% of assimilated energy in *Ilyanassa obsoleta* (Edwards and Welsh 1982). Thus there could be substantial costs from any increased secretion of mucus in keeping toxic chemicals out that are incurred for as long as exposure persists.

Other strategies for keeping toxic chemicals out may involve permanent changes in body surfaces (e.g., structural alterations in the cuticle of insects (Maynard Smith 1989)) that could in principle be detectably costly. Alternatively, keeping toxicants out might be achieved by retreating into an impermeable body cover such as a shell or tube and this could entail a reduction in metabolic rate but at the expense of reduced feeding (Newell 1979).

Cost of Pumping it Out

The active removal of toxicants is an inducible response, the cost of which will be incurred for as long as the chemical is present. The energetic cost of active transport is calculable but seemingly low relative to total metabolism (Potts and Parry 1964). Many observations of increased O_2 consumption of aquatic animals due to salinity stress were probably largely attributable to increased activity, but similar increases in isolated tissues may in part be due to enhanced active transport (reviewed by Newell 1979).

Cost of Detoxification

The capacity for detoxification can be inducible or constitutive; for example, detoxification enzyme activity might be consistently maintained at a high level or might only increase immediately following exposure. There are appreciable net energy costs to the processes of protein synthesis and possibly catabolism (Hawkins 1991) that are the basis for the induction of detoxification, excretion and repair processes (Calow 1989). It remains open to question, though, if in isolation the induction of specific protective proteins has an appreciable effect on energy budgets relative to the overall throughput of energy within organisms. For example, Barber et al. (1990) estimated metallothionein synthesis to represent <5% total metabolism in metal-challenged daphnids. However, even a small cost such as this might be appreciable, especially in terms of fitness, if experienced continuously or semicontinuously throughout the life cycle. Long-term costs may arise in association with physical alterations of the tissues or organs involved in detoxification. Fishes exposed to toxicants develop structural changes in their livers some of which are combative responses (Hinton and Lauren 1990). These are likely to be associated with increased costs related both to tissue construction and maintenance. Similar responses occur in other vertebrates (e.g., the liver weight of mammals increases in response to ethanol consumption (Rubin and Lieber 1974). However studies on the metabolic costs of developing and maintaining morphological structures are rare (Sibly and Calow 1986).

From metabolic rate measurements

A general prediction that arises from models involving graded physiological responses that are metabolically costly

(e.g., Selye 1950; Hatch 1962) is that metabolic rate should increase with increasing levels of toxicant (exposure time and/or concentration) until irreversible pathological effects impair metabolism itself (Calow 1989). There are some reports of this from observations of O_2 consumption (Basha et al. 1984; Le Bras 1987) and direct calorimetry (Gnaiger 1981). Aly (1993) measured oxygen consumption in chironomid larvae after increasing levels of whole-body gamma irradiation and observed first an increase and then a decrease in O2 consumption with increasing dose - consistent with predictions of a hump-shaped relationship between metabolic rate and chemical stress. Widdows and Donkin (1991) demonstrated that as tissue concentrations of pentachlorophenol in *Mytilus edulis* increased, respiration rate almost doubled with respect to control mussels, but then decreased to about half of the control rate at the highest tissue concentrations. Feeding rate began to decrease at the same tissue concentration at which respiration began to rise. Widdows and Page (1993) reported a similar respiratory pattern for mussels exposed to organotins. However, in this case feeding rates first reduced at concentrations an order of magnitude higher than those eliciting an increase in respiration, suggesting that "supply side" and "demand side" responses (see Baird et al. 1990) may vary, at least partly, independently.

In contrast to the results described in the preceeding paragraph, several reviews suggest that the relationship between metabolic rate and toxicant stress can take all possible forms, e.g., with metabolic rate increasing or decreasing continuously with the level of toxicant (e.g., Calabrese et al. 1977; McKim et al. 1987). The observation of such monotonic relationships may indicate real differences in toxic mode of action for different chemicals (e.g., Willows 1994, see below) or that individual researchers have, perhaps unintentionally, confined their measurements to one side or the other of the response curve. The shape of the metabolic response curve may be skewed to the left or right, increasing the chance that either the steep increase or steep decrease in response is missed by an experimental design that uses evenly-spaced test concentrations.

There are many problems in obtaining and interpreting direct measures of metabolic rate. Oxygen consumption is sensitive to a wide range of environmental conditions, apart from toxicants, and not all these have been properly controlled in experimental work. It is usual to recognize various components of total metabolism - basal, standard, active, etc. - and these need not all respond in the same way or synchronously to environmental stresses (Newell 1979). Some recent experiments on *Daphnia* demonstrated that though there was circumstantial evidence for the deployment of stress-resisting mechanisms (e.g., differential survival among genotypes) under chronic exposure to toxic chemicals, there was no significant increase in O_2 consumption (Baird et al. 1990; Barber et al. 1990) .

However, under these circumstances, there was a reduction in food intake and hence in the metabolic costs associated with this. Therefore, an unchanged metabolic rate here indicated that a greater proportion of total respiratory metabolism was concerned with maintenance.

Toxicants might uncouple oxygen consumption from phosphorylation (Slater 1963), and so increasing oxygen consumption might not be indicative of the rising costs predicted by metabolic models. Although oxygen consumption is probably the most widely used measure of metabolic rate, many organisms (esp. aquatic invertebrates) can have substantial anaerobic metabolic capacities and may engage in anaerobic metabolism even when oxygen is plentiful (Famme and Kofoed 1982). That such organisms can fuel some of their metabolic needs via anaerobic pathways may allow them to perform better than expected (on the basis of aerobic respiration alone), but may mean that they incur greater costs as well.

To summarize, there are a *priori* grounds for expecting physiological costs associated with processes deployed to cope with exposure to contaminants. There is some good, but not decisive, empirical evidence for increased metabolism associated with stress resistance. Further research is needed on the costs of particular processes, the careful measurement and partitioning of metabolic rate under stress, the identification of specific costs as inducible or constitutive and as short term or long term. Greater consideration should be given to the complications likely from the uncoupling of phosphorylation.

From energy budget studies.
Increased metabolic costs should appear as reductions in the production term of the energy budget equation, and in ecotoxicological studies this has often been referred to in terms of the scope for growth (SfG) concept of Warren and Davis (1967).

Energy Allocation Models
SfG has been observed to reduce under conditions of toxic stress for a variety of aquatic animals in both field and laboratory situations (e.g., Bayne et al. 1979; Naylor et al. 1989). However, care needs to be exercised here in that toxicants often impair energy intake ("supply side" effects) and this can dominate the metabolic response ("demand side" effects; but see Widdows and Page 1993). In other words, reduced SfG in response to chemical exposure may not always result primarily from increased metabolic cost.

Measurements of the components of SfG have been used to distinguish among different modes of toxicant action. Willows (1994) distinguished among toxicants acting by narcosis, metabolic inhibition and metabolic uncoupling by examining differential effects on respiration and feeding. The particular models that he employed predicted that the relationships between measured degrees of toxic stress and the maintenance ration required to maintain zero SfG may be highly nonlinear and dependent on the mode of toxicant action.

Heterozygosity and Maintenance Metabolism.
Low maintenance metabolic rates have been positively correlated with high SfG and with the level of genotype heterozygosity (Koehn and Bayne 1989). This has led to the idea that genotype-dependent maintenance metabolic rate can be used to predict the ability of organisms to withstand environmental stress. This view assumes that environmental stress increases metabolic cost for all individuals in a population, and this leads to the prediction that those individuals performing best (i.e., those that have the lowest maintenance metabolic rates) in optimal environmental conditions will be those that also perform best during exposure to toxic chemicals or other environmental stresses. However, the available empirical data relating genotype-controlled maintenance metabolic cost to susceptibility to chemical exposure remain equivocal (reviewed in Forbes, submitted).

From studies of relative fitness.
If resistance is costly, then an indirect consequence would be that in the absence of chemical contamination resistant genotypes should be selected against and the trait should disappear. This should be particularly the case for constitutive adaptations.

Relative Fitness in Stressed vs. Unstressed Environments
Studies of plants adapted to contaminated soils have found that tolerant individuals are competitively inferior to non-tolerant plants when grown on uncontaminated soil (e.g., Macnair, 1981). Certainly, heavy-metal tolerant strains of several species of flowering plant have lower production rates than non-tolerant ecotypes when grown in "clean" soil (Cook et al. 1972; Wilson 1988). Field data comparing some resistant and susceptible genetic strains of animals in stressed and favorable environments, indicate that whereas resistant strains are fitter under the appropriate stress, they are less fit in favorable environments (review Sibly and Calow 1989, their Table 1). Dieldrin-resistant genotypes of the sheep blowfly, *Lucilia cuprina,* appeared to suffer a disadvantage, relative to susceptible genotypes, in the absence of the insecticide (cited in Wood and Bishop 1981). Cadmium-tolerant populations of the midge,

Chironomus riparius, reared in the absence of cadmium, suffered high mortality and lowered larval growth rates and reproductive success (Postma et al. 1995). In this case, part of the increased 'cost' of cadmium tolerance appeared to be an increased physiological requirement for zinc.

The degree to which there are trade-offs between performance in chemically-stressful versus favorable conditions are likely to depend on the extent to which the stress-resisting processes involved are constitutive or inducible. If costs of resistance are only incurred during chemical exposure (i.e., responses are inducible), there would be no reason to expect a performance tradeoff. However, if resistance involves higher running costs at all times, resistant individuals would be at a fitness disadvantage in the absence of stress. Insecticide-resistant strains of *Drosophila melanogaster* have higher basal activities of various cytochrome P450-linked enzymes than susceptible strains, but the enzymes of the resistant strains are much less sensitive to induction by toxic chemicals (Fuchs et al. 1994). This suggests that chemical resistance in this system involves constitutive costs, and it would be very interesting to compare the relative fitnesses of these strains in the presence and absence of inducing chemicals.

Loss of Resistance

Studies of some, but not all, insecticide-resistant pest strains indicate a gradual loss of resistance in the absence of insecticide exposure (Bishop and Cook 1981). Data on reversion of DDT and dieldrin resistance in mosquitoes, following withdrawal of insecticides, were used to estimate selection coefficients against resistant genotypes (cited in Wood and Bishop 1981). The relative fitnesses of resistant homozygotes ranged from 0.44 to 0.97 compared to the susceptible homozygote (with a fitness of 1). The slight loss of resistance in cadmium-tolerant genotypes of the oligochaete, *Limnodrilus hoffmeisteri,* during two generations of laboratory culture in clean conditions may reflect the cost of cadmium tolerance (Klerks and Levinton 1989).

It is important to note that much of the data demonstrating the evolution and loss of resistance in chemically-exposed populations have been gathered in situations of extremely hard selection (i.e., high exposure concentrations often applied acutely) in which the genetic basis of resistance is often controlled by single genes of large effect (Wood and Bishop 1981). As the genetic basis of resistance to chronic, low exposure concentrations may involve many genes of small effect (Forbes and Forbes 1994, p. 193), the relationship between resistance and fitness may be more complex, and the costs of resistance may be difficult to estimate.

IMPLICATIONS

For individual and population performance

Here we consider first the implications of metabolic costs for the dynamics of populations as a whole before going on to consider the implications of these costs for different genotypes within populations and their consequences for evolutionary responses to chemical stress.

Consequences for Populations

Toxicants can directly impair survival, growth and reproduction, and these have obvious population level consequences. However, the costs of combating toxicants, by trading off with other elements of the energy budget that influence survival, growth and reproduction, can also influence population dynamics. In particular, there should be a trade-off between the capacity to survive toxic stress and growth rate and reproductive output (Sibly and Calow 1989). Of possible relevance in this regard are some recent studies of heat stress in *Drosophila melanogaster,* which demonstrated a clear trade-off between the induction of stress proteins (which was associated with increased survival during subsequent heat stress) and reduced fecundity (Krebs and Loeschcke 1994).

There are models that attempt to make explicit links between energy budget descriptions of individual physiology and the dynamics (i.e., change in abundance) of their populations (Kooijman et al. 1989; Nisbet et al. 1989; Hallam et al. 1990). These presume functional relationships between the energy allocation to reproduction (from the fraction of energy committed to producing propagules and their individual sizes) and the time between life-history events (from the fraction of energy committed to somatic growth and the form of growth curves), but have largely ignored the consequences for survival of investing in increased maintenance. The potential problems that arise from the latter are illustrated in the interpretation of SfG (above). This has frequently been used as a population-relevant index of stress, because reductions in it signal reductions in production, and hence fecundity and growth (and reductions in growth increase time between birth and breeding and between breeding periods). However, a reduction in SfG might be a consequence of increases in maintenance metabolism, due to the deployment of stress-resisting mechanisms. The consequent enhanced survival of individuals might compensate for reduced recruitment in terms of changes in abundance. The precise outcome depends upon the exact form of all the functional relationships involved (Calow and Sibly 1990) and these, therefore, deserve more thorough investigation.

There are other possible complications. For example, studies of the effects of the fungicide Calixin on two

earthworm species demonstrate that tradeoffs among life-history traits can vary among species and toxicants, as well as within a single species/toxicant combination during the time period following an exposure event (Mather and Christensen 1994). For one species, energy was allocated to adult growth, at the expense of the number and size of offspring during the initial month following fungicide exposure. However, over the next three months, energy was allocated to maintaining reproductive output, at the expense of adult growth and offspring size. The pattern described above differed for two other fungicides and the other species of earthworm studied.

Willows (1994) demonstrated another source of complexity in linking observations on SfG to population dynamics. He modeled the effects of toxicants, having known modes of action, on the components of the energy budget and found that equivalent effects on SfG could have very different consequences for the organism. He concluded that the relationship between toxicant concentration and the ability of an organism to maintain its SfG may be highly nonlinear and may be sensitive to e.g., food availability and the non-stressed metabolic rate.

In conclusion it is possible to make rigorous links between energy allocation patterns in individuals and the dynamics of populations, but these are complicated in various ways and, of course, do not take account of effects due to higher-order interactions through such processes as predation and competition.

Consequences for Selection Among Genotypes

Within populations, not all genotypes are likely to respond in the same way to toxic stresses. There are at least two possibilities: (1) different genotypes might occupy different positions on the trade-off curves between investment in combating toxic stress and production of biomass (e.g., perhaps because they are metabolically more efficient) and (2) different genotypes might vary in their ability to shift to different points on the trade-off curve depending upon conditions (i.e., due to differential ability to induce stress-resisting mechanisms).

Long-term exposure to chemical toxicants might favor either one of these kinds of evolutionary responses; the former (1) would lead to a constitutive adaptation - i.e., increased resistance at the expense of reduced production; the latter (2) would lead to an inducible adaptation - i.e., a capacity to "switch on" energy-expensive stress-resisting processes at appropriate times. Which evolves will depend upon the frequency of occurrence of the stress, and the genetic variance from which selection can be effected. Constitutive responses have been documented in flowering plants, but appear to be less obvious in animals where inducible responses have been documented (e.g., Baird et

al. 1990; Barber et al. 1990). It seems possible that this might be related to mobility; terrestrial plants are unable to escape from contaminated conditions whereas animals can move into and out of polluted patches. Although, as noted above, several strains of insecticide-resistant *Drosophila melanogaster* show a less chemically-inducible but relatively active P450 enzyme system compared to susceptible strains (Fuchs et al. 1994) and are therefore inconsistent with this hypothesis. This issue deserves further consideration.

The word 'might' was used above to qualify the prediction that chemical stresses will favor resistant genotypes, i.e., investment in the mechanisms that ameliorate the toxic effect of these chemicals. This is because there are two other alternatives. (1) It might pay to take the risk of being poisoned for the sake of growing rapidly (i.e., invest less, not more, in defense) so that the time between breeding events is reduced and hence the period over which mortality agents can act will, by definition, be reduced. Whether this will be favored depends, importantly, on the form of the trade-off curve and the way that itself responds to stress (Sibly and Calow 1989). Interestingly, it follows from this that we should expect more to be invested in general defense structures in more benign environments with high potential production than in more stringent environments with low production. This could explain the latitudinal increase in predator-defenses towards the equator in certain molluscs documented by Vermeij (1978) but not explained by him in this way (cf. Sibly and Calow 1989). (2) As well as, or possibly in place of, these direct evolutionary responses to stress there can also be indirect ones; i.e., impairment of one component of fitness by a toxic stress (e.g., growth rate) might favor other components of fitness (e.g., genotypes that start bigger). Such indirect, life-cycle responses have been documented in at least one population of aquatic invertebrates (Maltby et al. 1987).

In summary, exposure to contaminants can select for resistant genotypes even if resistance is associated with increased metabolic costs. However, several evolutionary outcomes are possible and it is difficult to predict a priori which one is the most likely.

For the community and ecosystem

At the community/ecosystem level there are also obvious consequences for costs associated with stress resistance. If these kinds of costs are incurred generally across species, then there will be an increase in observed R/B ratio, and a reduction in P/R and P/B ratios (where P = production; B = biomass; R = respiratory heat loss - all usually intended to apply to whole trophic levels). A further consequence of this is that the imbalance may make the

ecosystem more dependent upon energy sources from outside (Margalef 1975). Odum (1985) has already predicted these metabolic trends in ecosystems under stress, but on the basis of rather general thermodynamic arguments. However, it is likely to be difficult to move from observations on individual species to statements about the community or ecosystem as a whole because a) we do not know enough about the likely generality of the responses across species and b) there are likely to be complex relationships between species composition and the functional attributes of communities/ecosystems (Lawton 1994).

CONCLUSIONS

There are costs of living associated with maintaining biomass in an organized state (Prigogine et al., 1972). We have shown that these costs are likely to increase during toxic challenge, but that the extent to which this occurs remains open to question. If these costs are appreciable they will influence individual performance and may translate into decreased fitness. If the latter occurs there will be important implications for ecological and evolutionary processes. Hence, if increased metabolic costs are measured in response to chemical exposure, this suggests that there is an effect that might be ecologically relevant.

Low-level exposures to chemical contaminants, by definition, produce subtle responses in organisms that are usually measured at a suborganismal level. There has been a debate in ecotoxicology as to whether these kinds of responses are relevant at an ecological level and hence should be categorized as an 'effect' or 'no effect'. One possible criterion is whether the effects observed are associated with appreciable metabolic costs, for if no costs can be detected then demonstrating the implications for ecological-level processes becomes more problematic.

The cost hypothesis therefore has important implications for understanding mechanisms of response to toxicants at the individual level and their implications for ecological systems. However, there are a large number of complexities. These raise specific questions that, in principle, are open to further analysis through appropriate experimental programs. We see as the most important areas for further attention the explicit quantification of costs and the elucidation of the specific functional relationships between metabolic costs and life-cycle variables.

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