

AN ABSTRACT OF THE THESIS OF

Kelly M. Burnett for the degree of Master of Science in
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Title: The Multisteady-state Nature of Toxicant Behavior and Effect
in an Aquatic Laboratory Ecosystem

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The influence of competition, exploitation and level of energy input on toxicant behavior and effects in simple aquatic laboratory ecosystems was studied. Twenty-four systems composed of populations of guppies and snails and various taxa of algae were housed in 40-liter flow-through glass aquaria. The systems differed in organization and in level of environmental parameters. Guppies and snails were competitors for a common food resource. Systems were established in which guppies and snails were sympatric and allopatric. These systems were exposed to three different levels of guppy exploitation and received two levels of energy input. System dynamic and near steady-state behavior was documented through monthly measurements of population biomass. Biomasses of interacting populations were displayed on phase planes. Shifts in system structure followed exposure to a sublethal concentration of dieldrin¹. System responses ranged from system perturbation and recovery to guppy population extinction. Toxicant concentrations were determined in adult female fish from three systems under different conditions. Both toxicant effect and toxicant behavior were influenced by system organization and environment. Quite different conclusions concerning

toxicant performance could have been drawn if observation had been restricted to only one set of system conditions.

¹ dieldrin HEOD: endo exo isomer of 1,2,3,4,10,10-hexachloro-6,7-epoxy-1,4,4a,5,6,7,8,8a-octahydro-1,4,5,8-dimethanonaphthalene

Keywords: Competition, laboratory ecosystem, aquatic ecotoxicology
accumulation, exploitation, nutrient level

THE MULTISTEADY-STATE NATURE OF TOXICANT BEHAVIOR AND EFFECT
IN AN AQUATIC LABORATORY ECOSYSTEM

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INTRODUCTION

Compounds released by humans into the natural environment are no longer wards of a controlled laboratory. Their performances are subject to all biotic and abiotic influences exerted by the natural environment. Thus it becomes necessary to consider ecological characteristics of natural systems when approaching problems associated with toxicant distribution and effects.

Well-documented ecological influences upon toxic performances include those which exert their effect either directly upon the physiological processes of the organism or through physiochemical interaction with the toxicant itself. The influence of temperature on toxicant effects [1] or sediment type on toxicant availability [2] are established examples. Less obvious in effect yet probably of equal importance is the role that factors such as system productivity, exploitation by humans and species interactions such as competition or predation may play in shaping toxicant behavior and effects.

Laboratory ecosystems are often used in the pursuit of ecotoxicological understanding. They enable the systematic study of toxicant behavior and effects in relation to ecosystem performances including structure, organization, development and persistence over a range of environmental conditions. They also allow these system performances to be examined in view of their dynamic and multisteady-state behaviors. When conducted from a strong framework of conceptual and theoretical constructs, these laboratory ecosystem

studies may prove helpful in advancing understanding of toxicant behavior and effect in natural ecological systems [3,4].

To gain understanding of a system's capacity, system performance should be observed over a range of environmental conditions [4]. This is sometimes done in accumulation and simple toxicity studies. This approach is taken much less frequently in investigations of toxicant performances in ecosystems. In endeavoring to obtain an integrated insight into ecotoxicological interactions, only a few investigators have examined toxicant effects on laboratory ecosystems over a range of environmental conditions and system states [5,6,7]. This work was an attempt to advance understanding of toxicant behavior and effects in aquatic ecosystems. Toxicant performance was examined in laboratory ecosystems having different organizations and maintained under different levels of energy input and population exploitation.

Isocline Models of System Structure

Both theories and models are means by which we partially articulate, symbolize, and provide a perspective on relevant experience in a domain. Theories in their use of abstract language furnish a universal perspective; models make a contribution to explanation by providing a more particular tool to order experience.

Isocline models can be used to illustrate the joint influence of community organization and environment on toxicant effect [8]. Figure 1 represents a simple biological predation system. The relationships between populations composing this community and between these populations and their environment can be demonstrated with systems of

isoclines on phase planes (Figure 2). The intersecting isoclines define systems of steady-state relationships between the populations. The position and form of these isoclines are determined by systems of graphs or equations representing the rates of change in biomass of the component populations [8].

In the example in Figure 2, for each set of environmental conditions energy input (I) and exploitation rate (E), there exists a single two-dimensional steady-state point on each phase plane. For a given set of conditions, the set of these two-dimensional points, derived from all phase planes, defines the steady-state structure of the system. Under different levels of the controlling environmental parameters, the community will come to have different steady-state structures. Thus, depending upon the states of the environment, a system can have an infinite number of possible steady-state structures and, therefore, can be understood as a multisteady-state system.

Introduction of toxic substances can alter the structure and organization of systems. This is symbolized on phase planes by a shift in the location of steady state points, as shown in Figure 2.

On each phase plane, trajectories represent changes in densities of the populations composing the system. If environmental conditions are fixed, trajectories will converge upon steady-state points which represent steady-state system structure under those conditions. In systems where environmental conditions are rarely constant for long periods, i.e., most natural systems, trajectories can be understood as being in pursuit of constantly shifting steady state points [9].

MATERIALS AND METHODS

Twenty-four aquatic ecosystems, composed of populations of guppies (Poecilia reticulata), snails (family Planorbidae) and various taxa of algae, were established in the laboratory. Each system resided in a 40-liter glass aquarium adapted for flow-through usage. All aquaria received 200 ml/min. of heated well water at $21 \pm 0.5^\circ$ C. Dissolved oxygen was maintained at $8.2 \pm 0.5 \mu\text{m}/\text{l}$ and pH at 7.6-7.9. Systems were exposed to a 14 h light/10 h dark cycle. During the light cycle the tanks were exposed to 20.3 ± 2.3 quanta of light energy. Each tank was provided with approximately 3,450 cm³ of floating plastic plants available as cover for newborn guppies and snails. The primary source of energy input, an alfalfa ration (20% Oregon Test Diet and 80% alfalfa w/w [10, 11]), was introduced daily. Guppies and snails are thought to compete for the alfalfa ration with algae serving as an alternate food source for the snails [11].

The experimental design enabled evaluation of the influence of competition, system productivity and exploitation on toxicant behavior and effects. The experimental design is summarized in Table 1. Twelve systems were provided with 0.6 g of the alfalfa ration daily and 12 with 4.0 g. These are referred to respectively as low energy input (Low I) and high energy input (High I) systems. At each I, six systems maintained populations of guppies, snails, and algae (G-S-A systems) and six maintained only guppies and algae (G-A systems).

Every 28 days the systems were sampled and guppy populations exploited. Guppy populations were exploited at 0, 25, or 40% of

population biomass (wet weight) present at the end of the each sampling period (0E, 25E or 40E). A systematic exploitation schedule was developed for each exploitation rate [11]. Guppies were removed from aquaria with nets and anesthetized with MS 222. Length and weight were determined for harvested and unharvested fish. Population biomass and yield were calculated. Individual members of the snail population were counted and their shell diameters measured. Snails from each system were then weighed as a group. Snails and unharvested fish were returned to the tanks. Biomass data was used in construction of phase planes.

At four day intervals, accumulated sediments were siphoned from each system. Sediments were composed of uneaten alfalfa ration and feces which collected on aquaria floors. Organic sediment density was used as an index of uneaten alfalfa ration. To monitor changes in sediment density, samples were saved and dry weights determined twice monthly. Organic sediment density for system sampling dates were extrapolated from three-place running averages of bi-monthly sediment densities.

Continuous introduction of 0.5 ng/L dieldrin (Shell Technical Grade, 100% purity) into 17 of the laboratory systems began when the systems had established near steady-states (NSS). NSS structure was assumed when biomass trajectories of interacting populations fluctuated in a restricted phase plane region relative to previous fluctuations. The dieldrin concentration of 0.5 ng/L was considered sublethal (guppy 96 h LC 50, 21 ng/l [12]) and probably directly affected only guppies. Remaining NSS systems served as checks for changes in population response not associated with dieldrin.

The toxicant dilution and delivery system was similar to the continuous flow dilution apparatus described by Chadwick [13]. Water concentrations of dieldrin were determined weekly using the gas chromatographic procedures described by Shubat and Curtis [14]. Concentrations were corrected for 80% recovery.

Fish samples for dieldrin analysis were selected from systems which differed in organization and environmental conditions. Toxicant concentrations were determined in adult female fish from three systems that had established NSS, one system at Low I, G-S-A, OE and two systems at High I, G-A, 25E. Whole fish toxicant concentrations were determined using the extraction and gas chromatographic procedures of Shubat and Curtis [14] and were not corrected for $80 \pm 14\%$ recovery. Fish lipid content was determined during the extraction process.

RESULTS

NSS System Structure Prior to Dieldrin Introduction

A digraph of inferred system organization is given in Figure 3. Energy input (I) and exploitation rate (E) were the manipulated environmental parameters. At Low I, increased exploitation rate reduced NSS guppy biomass in both G-A and G-S-A systems (Figure 4). Competition appeared to be important in determining the NSS structures of Low I systems. The relationship between fish and sediment biomass was shifted to the left on the phase plane when the snail competitor was present. At each level of exploitation, both NSS guppy biomass and sediment density were reduced in the presence of snails (Figure 4). An inverse relationship between fish and snail biomass in G-S-A systems at Low I is shown in Figure 5. Reduction in NSS fish biomass brought about by increased exploitation rate resulted in increased NSS snail biomass. Prior to dieldrin introduction, fish populations were able to coexist with the snail competitor in all systems except those at the highest exploitation rate. Fish population extinctions occurred in both G-S-A systems at Low I, 40E (Figure 4, 5, Table 1). In G-A systems at Low I, 40E, fish population extinction occurred in only one system prior to dieldrin introduction.

Level of energy input (I) also affected NSS system structure. Increasing energy input from Low I to High I increased the NSS biomass of all system components. At High I, the entire domain of behavior for all systems was shifted to the right on the phase plane (Figure 6). As at Low I, increased E resulted in reduction of NSS fish

biomass in both G-A and G-S-A systems. At High I, the presence of the snail competitor was not associated with a leftward shift of the guppy-sediment relationship as observed on the Low I phase plane (Figure 4). However, at a given exploitation rate, NSS fish biomass was still lower in systems containing snails than in those without the snail competitor (Figure 6). Prior to dieldrin introduction, fish population extinction occurred in two system at High I. Both were 40E systems; one with and one without snails (Table 1). As at Low I, an inverse relationship between fish and snail biomass was observed at High I (Figure 7).

NSS System Structure Following Dieldrin Introduction

Shifts in system structure followed dieldrin introduction (Figures 4-7, solid symbols). These changes in system structure were seen as shifts in NSS behavior on phase planes which sometimes entailed guppy population extinction. System response to dieldrin was related to levels of I and E and to system organization. Inverse relationships between E and fish biomass and between fish and snail biomass and direct relationships between I and system structure observed prior to dieldrin introduction were maintained following introduction of the toxicant.

In all Low I systems, reductions in guppy biomass occurred after dieldrin introduction (Figures 4, 5). In systems not containing snails (Low I, G-A, Figure 4), fish populations at OE were capable of persisting at reduced NSS biomasses. Fish population extinction occurred in all G-A systems at 25E and 40E.

In Low I systems containing snails (Low I, G-S-A), fish populations at OE established new NSS densities below pre-dieldrin levels. Increases in NSS snail biomass accompanied these decreases in fish biomass (Figure 5). At 25E there was little shift in system structure; neither fish nor snail NSS biomasses changed appreciably following toxicant exposure (Figures 4, 5). Fish populations at 25E were able to persist throughout 13 months of continuous exposure to dieldrin while fish populations in their counterparts without snails (Low I, G-A, 25E) became extinct.

In High I systems without snails (High I, G-A, Figure 6) there were actual increases in NSS fish biomass at OE following dieldrin introduction. Of all systems exposed to dieldrin, these were the only ones in which fish populations established new NSS biomasses above pre-dieldrin levels. In G-A systems both at 25E and 40E, NSS guppy biomass decreased slightly. Following dieldrin exposure, the 40E fish population was capable of persisting, although at a reduced NSS biomass. In both Low I and High I systems, all other fish populations exploited at 40E were extinct by the termination of the experiment.

In High I systems with snails (High I, G-S-A), at OE, following initial reductions, fish biomass increased to overlap pre-dieldrin NSS levels. At 25E, NSS fish biomass was reduced. At both OE and 25E snail biomass increased following the reduction in guppy biomass (Figure 7). At 40E fish population extinction occurred after introduction of dieldrin.

NSS Toxicant Behavior

In each of the three systems analyzed, dieldrin content increased with increasing fish body weight (Figure 8). Systems were compared based on analysis of variance of regression coefficients. At Low I, G-S-A, OE, fish of a given body weight contained greater quantities of dieldrin than fish from High I, G-A, 25E systems ($P < 0.05$). Furthermore, fish of a given body weight from the two High I, G-A, 25E systems contained similar quantities of dieldrin ($P > 0.05$). As dieldrin is a highly lipophilic compound, its content should be proportional to the lipid content within an organism. To determine if the observed pattern of results was simply a function of differences between systems in lipid content of fish, guppy whole body dieldrin contents were compared on a lipid basis (Figure 9). Differences in dieldrin content remained; fish from Low I, G-S-A, OE systems maintained greater dieldrin concentrations per unit of lipid weight than fish from High I, G-A, 25E systems ($P < 0.05$). The similarities in dieldrin concentrations of fish from the two High I, G-A, 25E systems were also still apparent ($P > 0.05$).

DISCUSSION

The laboratory ecosystems could be understood as multisteady-state systems [9]. More complex laboratory ecosystems containing amphipods and various microinvertebrates as well as guppies and snails have also been shown to exhibit multisteady-state behavior [7, 8]. Prior to dieldrin introduction, NSS system structure was determined by exploitation rate, energy input level and presence of the snail competitor. Introduction of dieldrin brought about shifts in system structure. Differences in dieldrin's behavior and effect in these systems are in part related to differences in system organization and environmental conditions. Although exploitation rate, level of energy input and competition undoubtedly interactively influence toxicant behavior and effect, it is tempting to formulate discussions which focus upon each factor's independent influence. If done thoughtfully, generalizations, useful in increasing understanding of toxicant performance in these laboratory systems, may be made.

Heavily exploited systems appeared to be more vulnerable to dieldrin exposure than systems exploited at lower rates or unexploited systems. Following dieldrin exposure, fish population extinctions occurred only in systems where the fish population was exploited (Table 1). A similar relationship between exploitation rate and population extinction was found in larger, more complex laboratory ecosystems [7, 8]. Regier and Loftus [15] related decline in fish populations in the Great Lakes to both eutrophication and overharvesting. They suggested that harvesting rate of Great Lakes

fishes should have been reduced to help compensate for increased stress caused by eutrophication.

System response to dieldrin was related to level of system productivity. Systems at High I appeared less susceptible to dieldrin's toxic effect than those at Low I. Following dieldrin-exposure there were fewer fish population extinctions at High I than at Low I. In addition, only High I systems showed any increase in biomass following dieldrin introduction. Post-dieldrin NSS fish biomass in High I, OE, systems without snails increased to above pre-dieldrin levels. Following initial reductions, fish biomass in High I, OE, systems containing snails increased to overlap pre-dieldrin levels. In a similar investigation with a more complex laboratory ecosystem, dieldrin's effect also appeared most pronounced at the lower energy input rate [8]. A relationship between nutrient level and toxicity has been observed in laboratory ecosystem studies of metal toxicity [3, 16]. The greatest alteration of net photosynthetic response in systems exposed to 10 ppm Cd was found in phosphorous limited systems [3]. Sugiura et al. [16] suggests that the characteristic response of a system to chemical stress varies with the nutrient level and that this should be considered when attempting to identify toxicity potential in natural ecosystems.

Competition also played a role in determining the effect of dieldrin in the laboratory systems. The exact nature of this role is by no means clear. Liss et al. [5] suggested that in the presence of a competitor, a predator population could be rendered more sensitive to a toxicant and more vulnerable to extinction. Experimental results

obtained under all but one system condition tend to support this assumption. At Low I, 25E, after introduction of dieldrin, fish population extinction occurred only in systems in which the snail competitor was absent. No explanation for this is immediately apparent. There was, however, a disparity in age and size distributions between G-S-A and G-A systems at Low I, 25E which may have been a contributing factor.

Just as competition between fish and snail populations influenced dieldrin's effect upon fish populations, competition also affected snail populations. The direct effects of the toxicant on guppies indirectly affected the snail competitor. At both Low I and High I, at OE and 25E, reductions in fish biomass brought about by the toxicant resulted in increased snail biomass.

Species interactions have been shown to shape toxicant effects in laboratory studies. The physical activity of the mayfly nymph, Hexagenia, was shown to intensify the toxic response of Daphnia held in water above contaminated sediments (17). Cooney et al. [1] in comparing the toxicity of acridine to stock and fed cultures of the calanoid copepod Diaptomus clavipes Schacht, suggested that intraspecific competition appeared to increase toxicity of the stock cultures at 16° C.

For the purpose of simplification, this discussion attempted to develop generalizations concerning the roles exploitation rate, level of energy input and competition had in influencing toxicant performance. It must be remembered that the NSS structures which were established, including fish population extinctions which occurred,

were a result of all environmental and organizational factors operating together in these systems.

Like toxicant effects, toxicant behavior may be influenced by organization and environmental conditions. NSS dieldrin concentrations were different in fish from systems that differed in organization and environment and were similar in fish from systems having similar organization and environment. Environmental factors affecting the amount of lipid in an organism have been shown to influence accumulation of lipophilic compounds [14, 18]. However, differences in quantity of lipid could only partially explain the observed pattern of NSS dieldrin concentrations in fish from our laboratory ecosystems. When concentrations were expressed as dieldrin per gram lipid, the relationships between systems remained. Shubat and Curtis [13] proposed dieldrin's accumulation in rainbow trout was connected with the composition of lipid constituents in tissues and that qualitative changes in lipid could result from ration level and/or dieldrin treatment. Although we did not attempt qualitative analysis of lipid constituents in fish tissues, Shubat and Curtis' proposal provides a way of interpreting the observed pattern of dieldrin accumulation which implies a relationship between dieldrin's behavior and system conditions.

Results from this study emphasize that a comprehensive examination of toxicant behavior and effects over a full range of system environmental and organizational conditions is essential for a thorough characterization of system performance. Had dieldrin's performance in these systems been examined under only one set of

system conditions, quite different conclusions could have been drawn. If the goal of investigation is to understand toxicant performance in a system, then methods which examine these performances over a range of environmental parameters and system organizations are necessary.

Table 1. Species composition and environmental parameters for the laboratory systems. Guppy population extinctions occurring prior to and after dieldrin introduction are given. Dieldrin was not introduced into systems whose guppy populations became extinct prior to the time of toxicant introduction.

Energy ¹ input rate	System ²	Exploitation ³ rate	Number of tanks	Pre-dieldrin guppy extinctions	Tanks receiving dieldrin	Post-dieldrin extinctions
Low I	G-A	0%	2	0	2	0
		25%	2	0	2	2
		40%	2	1 ⁴	1	1
	G-S-A	0%	2	0	2	0
		25%	2	0	2	0
		40%	2 ⁵	2	0	0
High I	G-A	0%	2	0	2	0
		25%	2	0	2	0
		40%	2	1 ⁴	1	0
	G-S-A	0%	2 ⁵	0	1	0
		25%	2 ⁵	0	1	0
		40%	2	1 ⁴	1	1

¹ High I = 4.0 g, Low I = 0.6 g alfalfa ration daily

² G = guppy, S = snail, A = algae

³ % of total guppy biomass removed during monthly sampling (w/w)

⁴ Not included on phase plane representation

⁵ One system was not shown on phase planes and served as check of system changes not associated with dieldrin

Figure 1. A. Diagraph of a simple predation system. Populations of a carnivore (C), herbivore (H), plant (P), and plant resources (R) define the system. Carnivore exploitation rate (E), rate of energy and material input (I), and toxicant concentration (T) define the environment of the system. Arrows indicate direct interactions between the populations and between these populations and their environments.

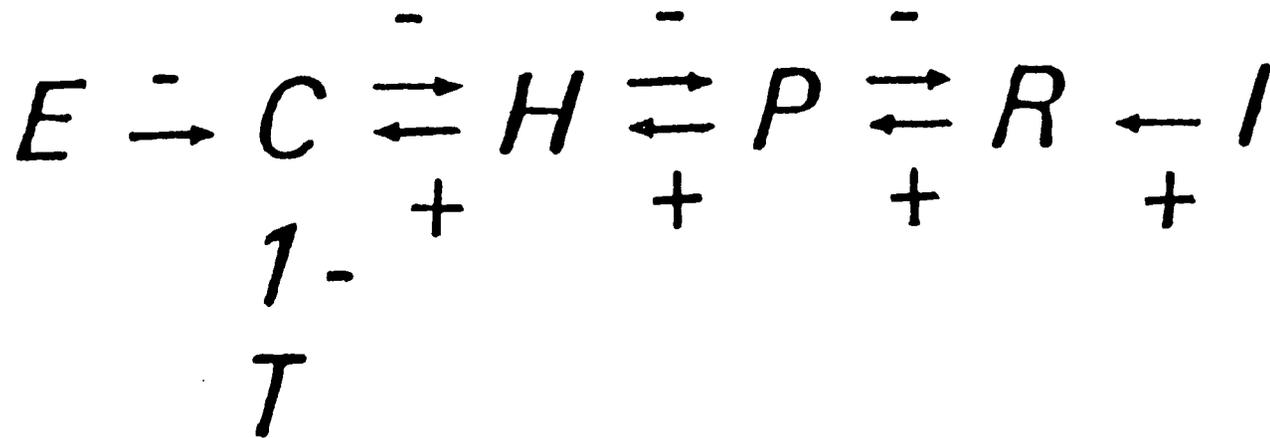


Figure 1.

Figure 2. Phase planes and isocline systems modeling the interrelationships between populations in a system. Where C, H, P, and R comprise the system and E, units of harvesting effort, and I, rate of input of plant resources, are system environmental parameters. A toxicant directly affects only the carnivore population. On each phase plane predator biomass is plotted on the y-axis and prey biomass is plotted on the x-axis. The descending lines identified by different rates of plant resource input, I, are prey isoclines. Each prey isocline is defined as a set of biomasses of predator and prey where the rate of change of prey biomass with time is zero. The ascending lines on each phase plane are predator isoclines. Each predator isocline is defined as a set of biomasses of predator and prey where the rate of change of predator biomass with time is zero. Each intersection of a predator and prey isocline is a steady-state point where the rate of change of both predator and prey biomass with time is zero. The presence of the toxicant lowers the predator isocline at each E on the C-H phase plane, the extent to which it is lowered depends upon the effect of the particular toxicant concentration on carnivore growth, reproduction and survival. Steady-state system structure at High I, OE, OT (circles); High I, OE, 2T (squares); Low I, 90E, 2T (triangles) is shown. Trajectories of biomasses of carnivore (C), herbivore (H), plant (P), and plant resource (R) originating at point 0 are shown to converge on each of these steady-states under each particular set of environmental conditions. After [8].

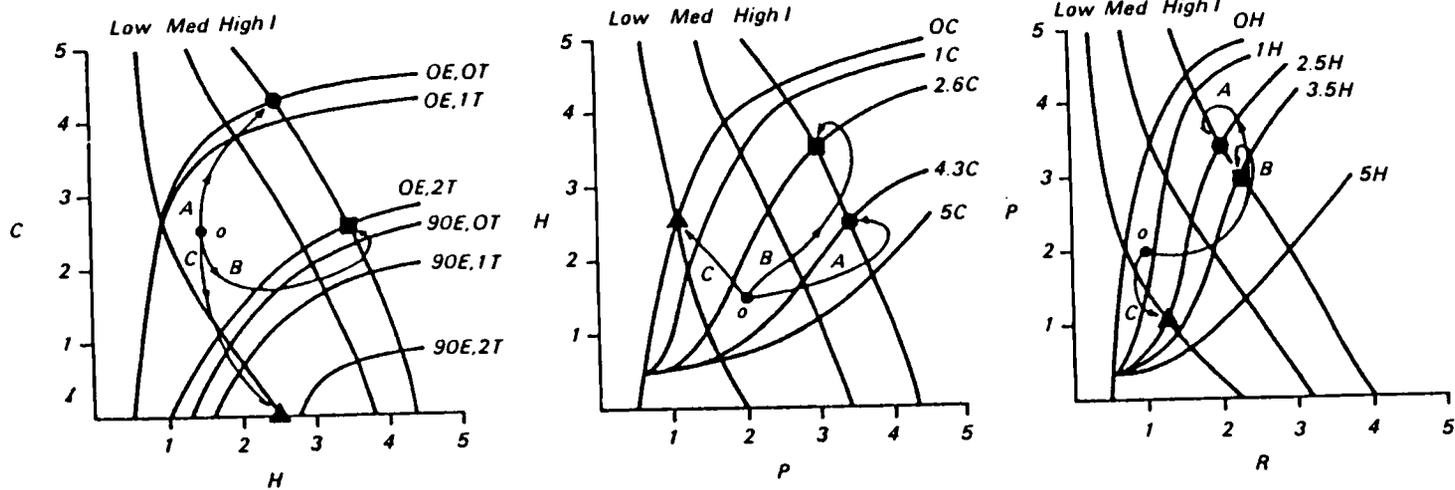


Figure 2.

Figure 3. Diagraph of the inferred trophic relationships in the experimental system prior to dieldrin introduction. The experimental system is composed of a guppy population (G), a snail population (S), an algal assemblage (A), and an alfalfa ration (AL). E is the exploitation rate to which the guppy population is exposed, expressed as percent population biomass removed every 28 days. I (alfalfa) and I(light) are the energy and material input rates of alfalfa and light into the system. E, I (Al), and I (light) may be taken to define the environment of the system. After [11].

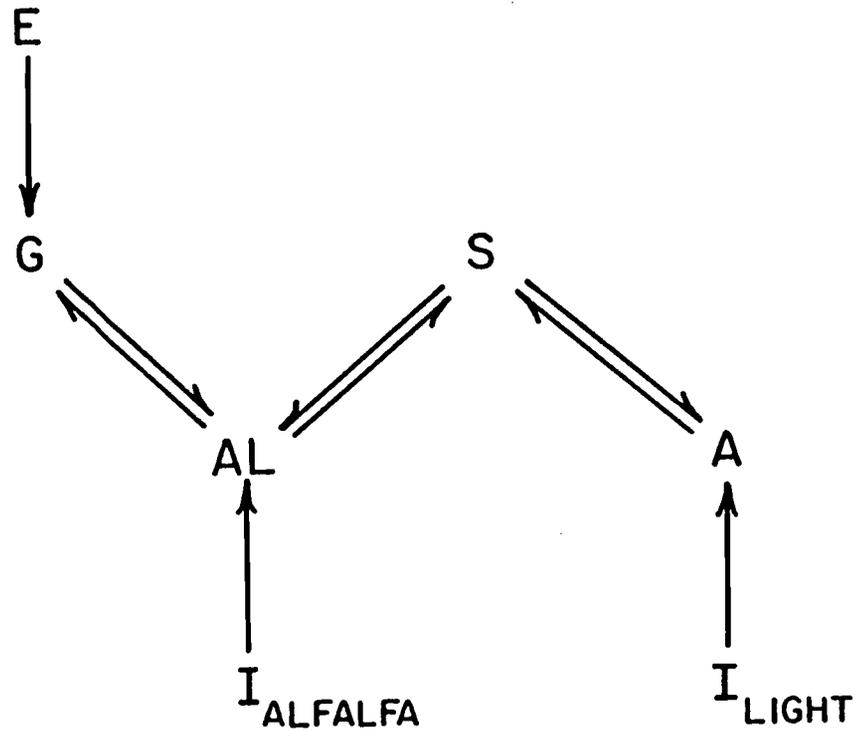


Figure 3.

Figure 4. Fish-organic sediment phase plane at Low I for systems in which snails are absent (G-A systems) and in which they are present (G-S-A systems). At each E, NSS behavior prior to dieldrin introduction is indicated by open symbols and behavior after dieldrin introduction is indicated by solid symbols. Only the means of NSS points (large symbols) are shown for one G-A system at OE, two G-A systems at 25E prior to dieldrin introduction, one G-A system at 40E prior to dieldrin introduction, one G-S-A system at OE, and two G-S-A systems at 40E. The number identifies the first NSS point.

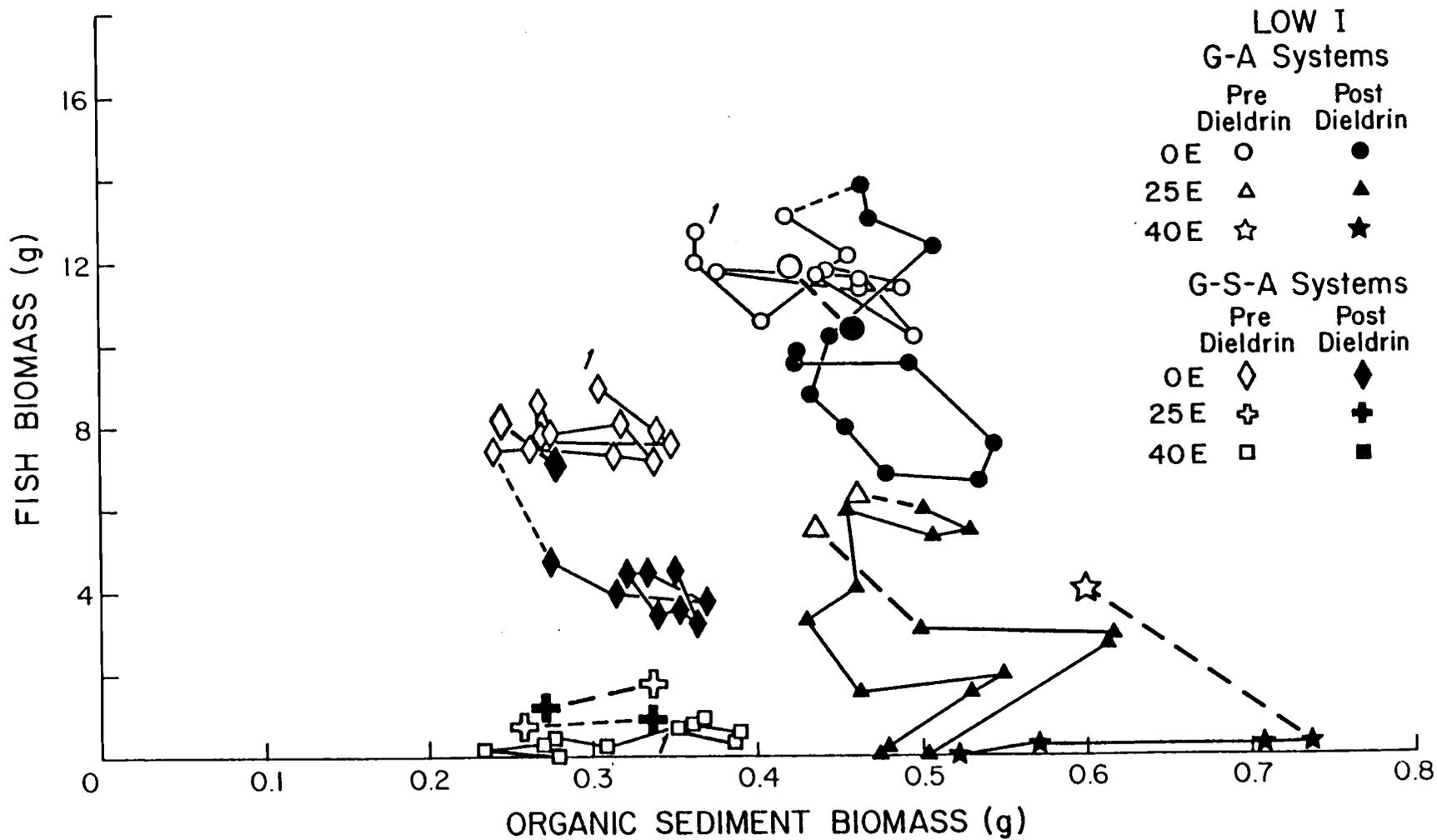


Figure 4.

Figure 5. Fish-snail phase plane at Low I. At each E, NSS behavior prior to dieldrin introduction is indicated by open symbols and behavior after dieldrin introduction is indicated by solid symbols. The number identifies the first NSS point. For selected systems, trajectories of points define dynamic and NSS behavior. Only the means of NSS points (large symbols) are shown for one system at OE and one at 25E.

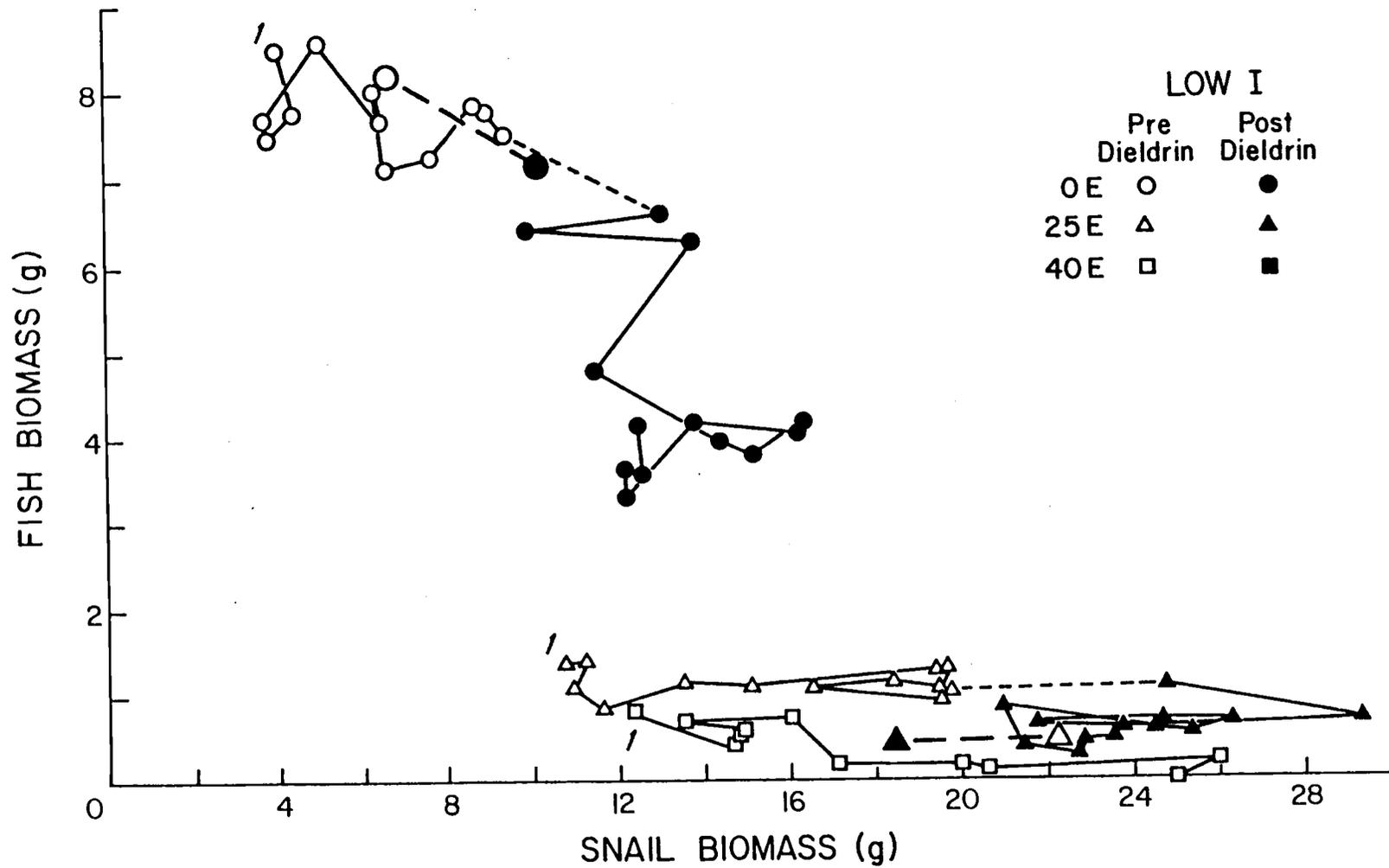


Figure 5.

Figure 6. Fish-organic sediment phase plane at High I for systems in which snails are absent (G-A systems) and in which they are present (G-S-A systems). At each E, NSS behavior prior to dieldrin introduction is indicated by open symbols and behavior after dieldrin introduction is indicated by solid symbols. Only the means of NSS points (large symbols) are shown for one G-A system at OE and at 40E, two G-A systems at 25E, and the G-S-A system at 40E prior to dieldrin introduction. The number identifies the first NSS point.

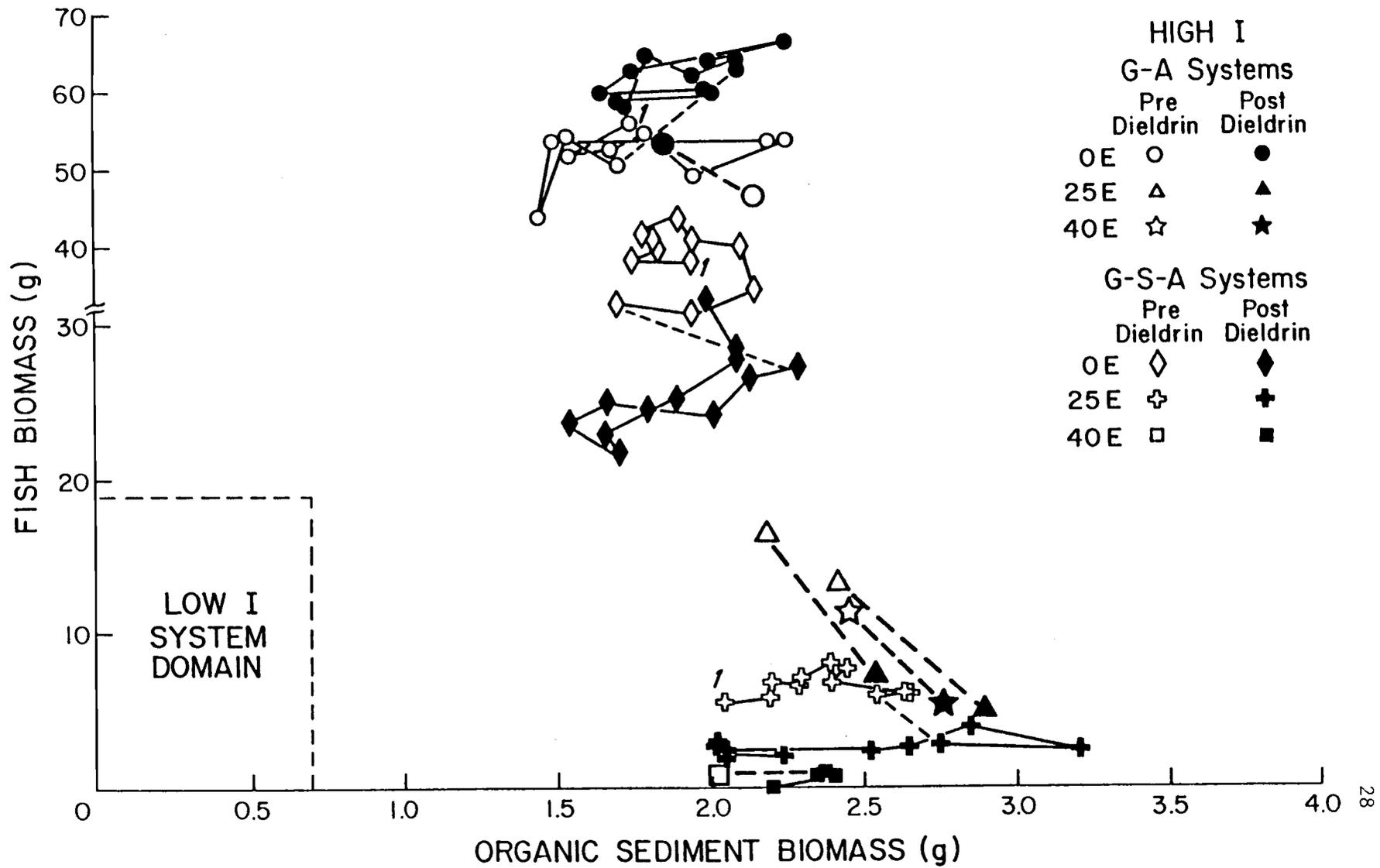


Figure 6.

Figure 7. Fish-snail phase plane at High I. At each E, NSS behavior prior to dieldrin introduction is indicated by open symbols and behavior after dieldrin introduction is indicated by solid symbols. The number identifies the first NSS point.

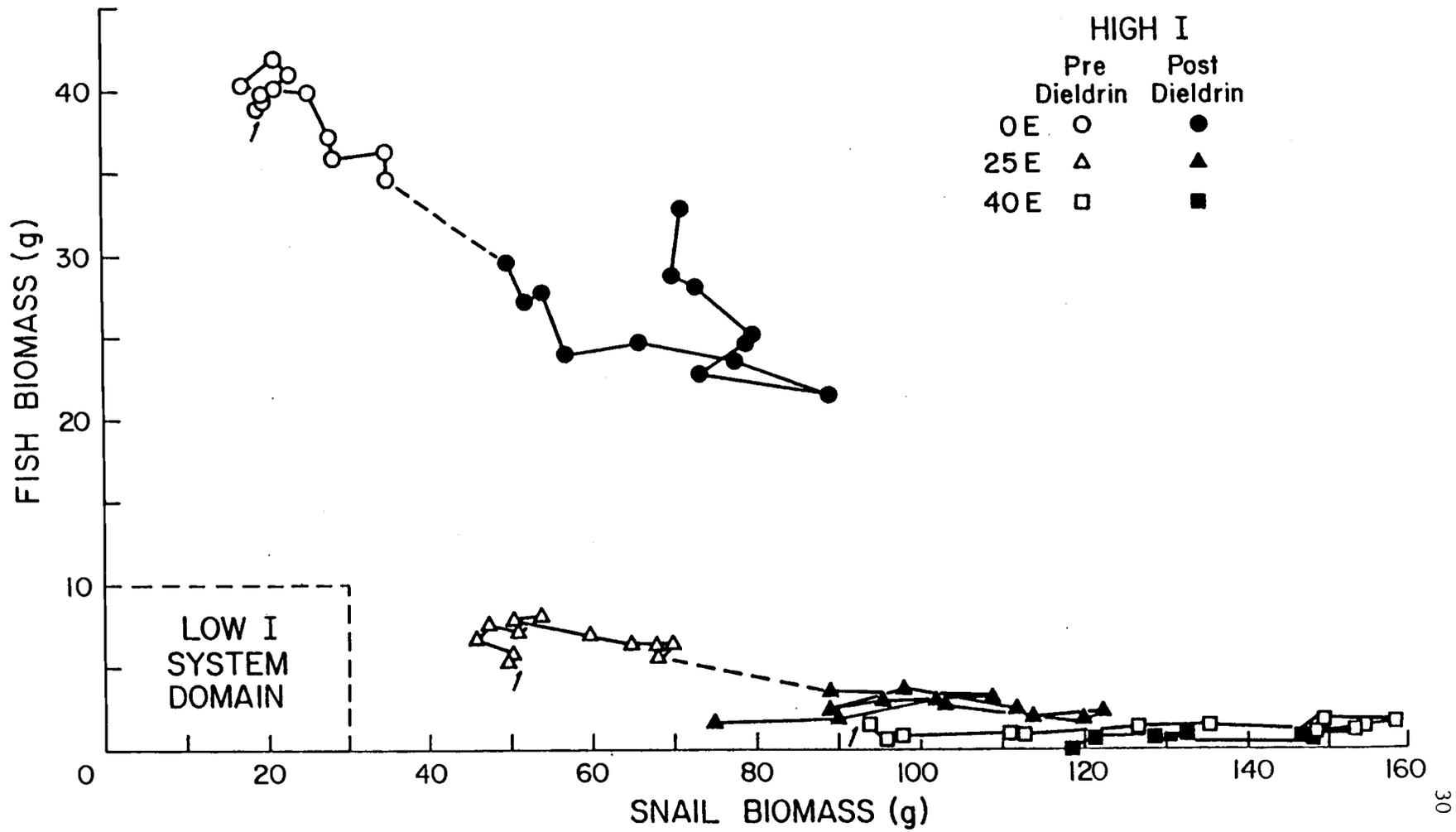


Figure 7.

Figure 8. Relationship between amount dieldrin in the whole body of guppies and guppy body weight at NSS for fish from one system at Low I, G-S-A, 0E (open triangle: $y = (2.39 \cdot 10^5) X - 3960$, $r^2 = 0.939$) and fish from two systems at High I, G-A, 25E (open squares: $y = (1.17 \cdot 10^5) X - 735$, $r^2 = 0.994$ and solid circles: $y = (1.31 \cdot 10^5) X - 1680$, $r^2 = 0.971$).

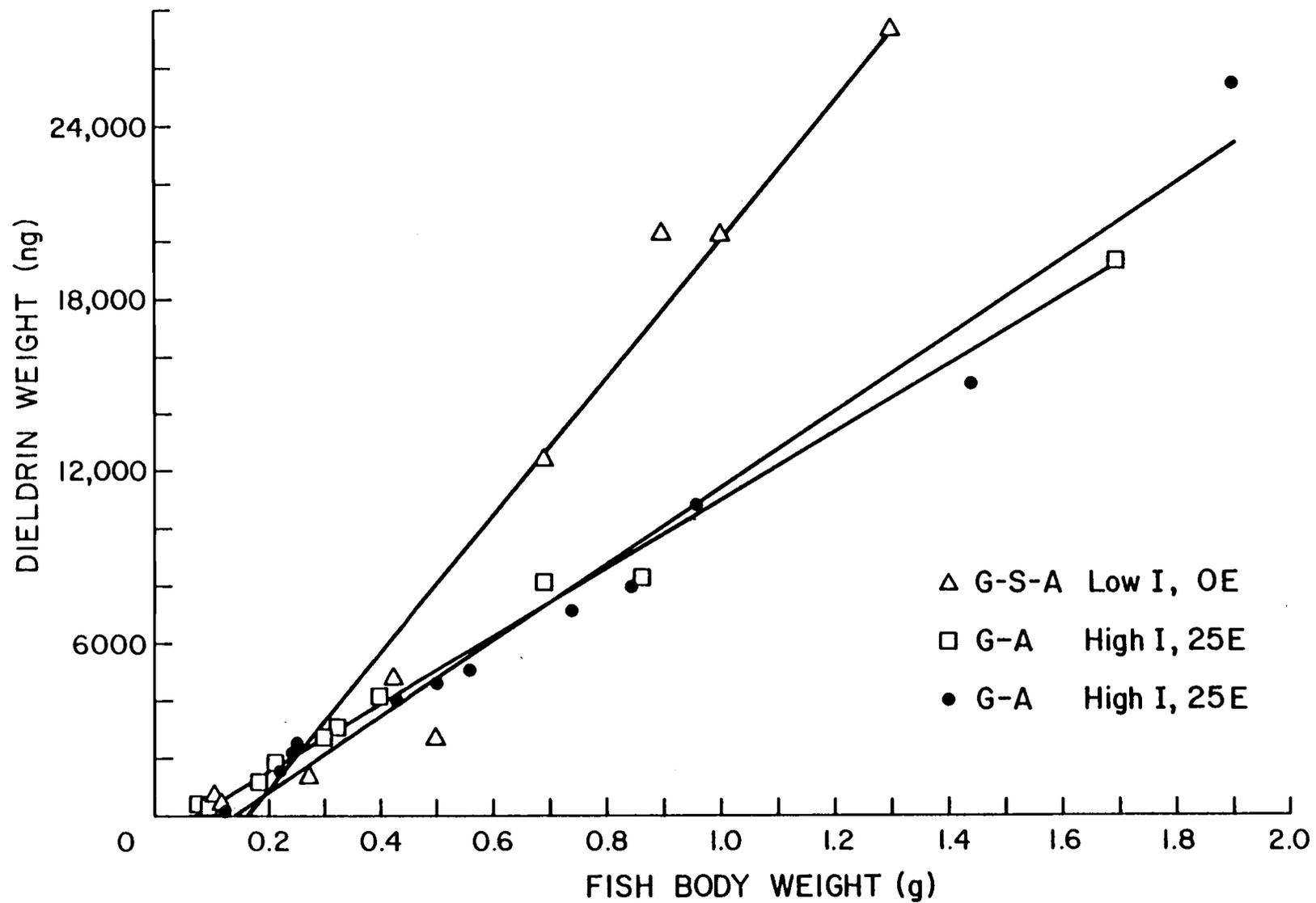


Figure 8.

Figure 9. Relationship between amount dieldrin in the whole body of guppies and total lipid weight at NSS for fish from one system at Low I, G-S-A, OE (open triangle: $y = (1.93 \cdot 10^5) X + 352$, $r^2 = 0.992$) and fish from two systems at High I, G-A, 25E (open squares: $y = (1.51 \cdot 10^5) X + 101$, $r^2 = 0.992$ and solid circles: $y = (1.48 \cdot 10^5) X - 299$, $r^2 = 0.986$).

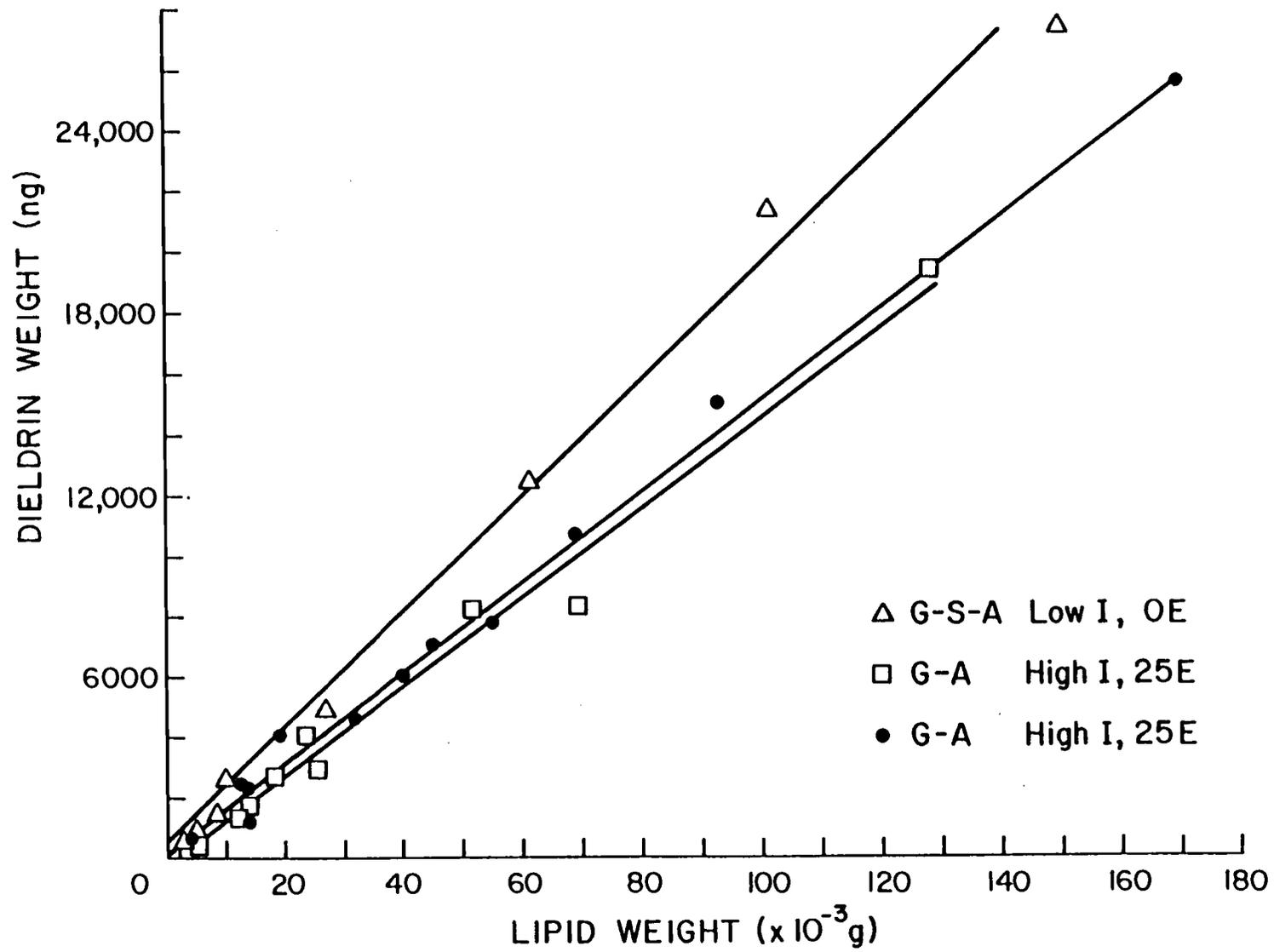


Figure 9.

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