

1 Paleo-ecotoxicology: what can lake sediments tell us about ecosystem responses to
2 environmental pollutants?

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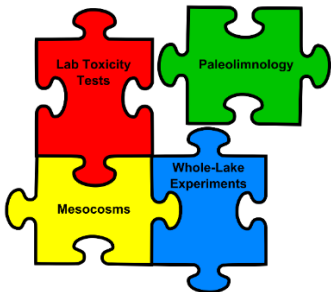
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18 **Abstract**

19 The development of effective risk reduction strategies for aquatic pollutants requires a
20 comprehensive understanding of toxic impacts on ecosystems. Classical toxicological studies are
21 effective for characterizing pollutant impacts on biota in a controlled, simplified environment.
22 Nonetheless, it is well-acknowledged that predictions based on the results of these studies must
23 be tested over the long-term in a natural ecosystem setting, to account for increased complexity
24 and multiple stressors. Paleolimnology (the study of lake sediment cores to reconstruct
25 environmental change) can address many key knowledge gaps. When used as part of a weight-
26 of-evidence framework with more traditional approaches in ecotoxicology, it can facilitate rapid
27 advances in our understanding of the chronic effects of pollutants on ecosystems in an
28 environmentally realistic, multi-stressor context. Paleolimnology played a central role in the
29 Acid Rain debates, as it was instrumental in demonstrating industrial emissions caused
30 acidification of lakes and associated ecosystem-wide impacts. “Resurrection Ecology” ”
31 (hatching dormant resting eggs deposited in the past) records evolutionary responses of
32 populations to chronic pollutant exposure. With recent technological advances (e.g.
33 geochemistry, genomic approaches), combined with an emerging paleo-ecotoxicological
34 framework that leverages strengths across multiple disciplines, paleolimnology will continue to
35 provide valuable insights into the most pressing questions in ecotoxicology.



37 **Introduction**

38 Water and sediment quality guidelines for several common pollutants have been
39 established for many regulatory frameworks, with the goal of balancing economic gains from
40 industrial activities against the potential for ecological harm in freshwater ecosystems. Such
41 guidelines are intended to provide a scientific basis for setting appropriate targets for the
42 protection of aquatic life and ecosystem services, and are typically reported in environmental
43 assessments. However, water and sediment quality guidelines are based largely on laboratory
44 bioassays using a limited number of taxa as model organisms [1- 2]. Emerging –omics
45 approaches (e.g. proteomics, metabolomics) provide further insights on the mode of toxic action
46 and sublethal effects of contaminants on metabolic pathways, gene expression, endocrine
47 functioning [3-5]. Syntheses of these studies are then used to estimate the concentrations above
48 which we would expect a pollutant to have deleterious ecological effects [6].

49 It is well acknowledged, however, that classical lab toxicity tests do not necessarily
50 reflect what is likely to occur in a natural ecosystem setting. For example, lab bioassays often use
51 standardized aqueous media, which does not capture the full range of variability in water
52 chemistry conditions likely to be encountered in the environment. The bioavailability and
53 toxicity of pollutants in aquatic ecosystems can be highly dependent on physicochemical
54 parameters like pH, alkalinity, and dissolved organic carbon [7-9]. In addition, only a small
55 number of species are used routinely for laboratory toxicity testing, to allow for ease of
56 comparison among studies. Care must be taken when extrapolating results based on a few
57 species to whole communities and ecosystems, as sometimes even closely related species may
58 have very different ecological tolerances [10-11]. There is also increased complexity in a natural
59 ecosystem compared to a controlled laboratory setting due to ecological interactions (e.g.

60 competition, food web interactions) [12] and confounding stressors (e.g. climate change) [13].
61 This may result in organisms being more sensitive to pollutants in a natural ecosystem setting
62 than would otherwise be expected based purely on lab bioassays [14]. Finally, lab toxicity testing
63 is usually done on short timescales, as even chronic toxicity tests typically run for less than a
64 month [15-16]. Sublethal effects [17] that can cause ecological harm to a population over the
65 long-term may not be adequately captured over these short timescales, nor are complex
66 ecological interactions readily observable in lab experiments. For example, a multi-year whole-
67 ecosystem acidification experiment conducted at the Experimental Lakes Area in northwestern
68 Ontario, Canada, showed that trout stopped reproducing at pH values higher than was anticipated
69 based on lab toxicity experiments, which was the result of starvation following the loss of acid-
70 sensitive prey species [18-19].

71 Controlled laboratory bioassays are clearly needed to establish the direct toxicity of
72 pollutants on aquatic biota in the absence of confounding factors but, for the reasons stated
73 above, the conclusions drawn from these studies need to be rigorously tested in a natural
74 ecosystem setting. Mesocosm experiments conducted *in-situ* in a lake or stream ameliorate some
75 of the limitations of lab experiments by simulating (albeit in a simplified way) the basic elements
76 of an ecosystem [20-21]. Field surveys that characterize aquatic communities and biodiversity
77 across broad environmental gradients can also be used to make inferences about the
78 concentrations of pollutants required to cause deleterious ecological effects. Whole-ecosystem
79 manipulation experiments, although logistically intensive and expensive, more fully capture the
80 complexity of ecosystem responses to controlled inputs of a pollutant [22-23]. Each of the
81 above-described approaches in aquatic ecotoxicology has strengths and limitations that, when
82 used in a weight-of-evidence approach, can provide a more holistic picture of the potential for

83 ecological harm from pollutant exposure. However, each of these approaches are still conducted
84 on relatively short timescales (hours to a few years) due to logistical constraints, when a longer-
85 term perspective is critical for characterizing the full range of natural variability needed to
86 contextualize complex and nuanced responses of ecosystems to pollutants [24]. We have argued
87 for the development of a new research framework that better integrates the principles of
88 ecotoxicology with the use of lake or other sediment cores as natural archives of environmental
89 change (the field of paleolimnology), providing the critical long-term perspective that is largely
90 missing in ecotoxicology [25]. The purpose of this review is to document specific examples
91 where paleolimnology has already contributed to advancing aquatic ecotoxicology, and discuss
92 new opportunities to utilize lake sediment core records to directly test conclusions drawn from
93 lab/mesocosm experiments in natural ecosystems, over long timescales. We also provide
94 recommendations for future advances in the emerging sub-discipline of “paleo-ecotoxicology.”

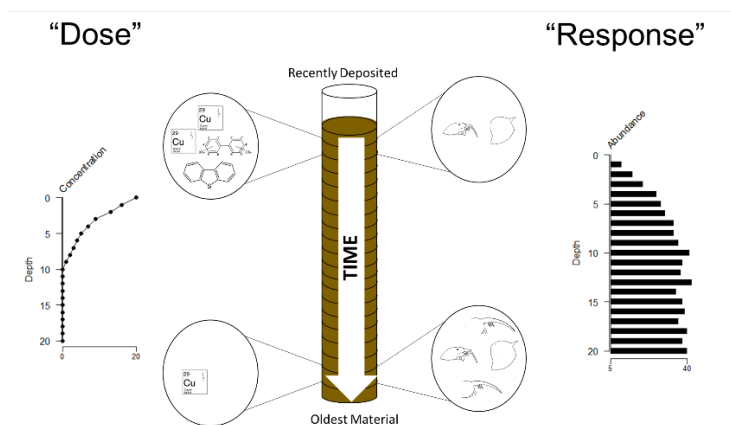
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96 **Contributions of paleolimnology towards our understanding of the movement and fate of** 97 **contaminants in the environment**

98 Lake sediments are partial sinks for many metals and persistent organic pollutants, and
99 the use of paleolimnological approaches to study the movement of contaminants in the
100 environment is well established [26]. Key questions such as “what is the doubling rate of
101 chemicals in the environment,” and “how long do they persist in the environment after bans or
102 restrictions on production are enacted” could not have been adequately answered without the use
103 of paleolimnological techniques [27-29]. For contaminants that have both natural and industrial
104 sources, such as metals and polycyclic aromatic compounds (PACs), paleolimnology often
105 provides the only means available to establish whether elevated contaminant burdens in a lake

106 are due to industrial activities or are naturally occurring [30-31]. Paleolimnological studies have
107 also been instrumental for understanding the movement of contaminants at a global scale, for
108 example by providing evidence for the influence of the global fractionation hypothesis on
109 latitudinal trends in contaminant histories [32-33], as well as the role biovectors like migrating
110 seabirds and anadromous salmon play in concentrating pollutants in lakes [34-35]. With recent
111 advances in methodology for studying post-depositional processes and source discrimination,
112 paleolimnological research continues to generate new knowledge on the fate of contaminants in
113 the environment [26].

114 Paleolimnology is not only useful for tracking contaminant deposition histories, but can
115 also provide new information on the ecological consequences of long-term exposure to
116 contaminants (“paleo-ecotoxicology”). Well-developed techniques are available for inferring
117 both historic contaminant inputs (the stressor record) and concomitant changes in biological
118 communities over time (the response variable) for several trophic levels across multiple habitats
119 (benthic, littoral, pelagic), including many of the common model organisms in ecotoxicology
120 (Figure 1).



121
122 **Figure 1.** Lake sediment cores are routinely used to infer historic depositional trends for many environmental
123 contaminants of interest, including metals and persistent organic pollutants (the “dose” or “stressor” record). Lake
124 sediment cores also preserve the fossil remains of many different groups of biological organisms (the “response”
125 record), including taxa routinely used as model organisms in ecotoxicology, such as *Daphnia* and *Bosmina* (shown
126 as drawings of commonly recovered fossil remains).

127 Considerable advances have been made over the last few decades on our ability to
128 recover high-resolution sediment cores that can be sectioned and dated (e.g. ^{210}Pb
129 geochronology) to provide reconstructions of past pollution and ecosystem trajectories within a
130 multiple-stressor context [36]. Importantly, sediment cores represent material that is temporally
131 integrated (typically 2-10 years per 1-cm interval), reducing environmental noise from seasonal
132 and inter-annual variability, and thus provide a clear indication of when significant ecological
133 thresholds are crossed in response to a stressor. Merging paleolimnology with ecotoxicology can
134 make significant, novel contributions to our understanding of the legacy of industrial chemicals
135 in the environment and their effects on our ecosystems. With the exception of a few notable
136 examples (which we will discuss), this is not commonly done, and represents a fruitful area for
137 future research.

138

139 **Biological organisms that can be studied from sediment cores**

140 Many different groups of biological organisms leave some form of identifiable subfossil
141 remain preserved in the sediment record [37-38]. Here, we focus on the most common biological
142 proxies used in paleolimnological studies, with an emphasis on those that are especially relevant
143 to ecotoxicologists.

144

145 *Diatoms*

146 Siliceous diatom frustules (cell walls) are well preserved in lake sediments. Historically,
147 they are the most commonly used biological proxy in paleolimnological studies, used to
148 quantitatively reconstruct changes in lakewater pH [39] and nutrients [40-41], as well as
149 qualitatively assess changes in other limnological properties like dissolved organic carbon

150 (DOC) [42], and lake thermal and ice-cover regimes in response to climate warming [43].
151 Several studies have used diatoms in lake sediment cores to track metal contamination in mining-
152 impacted regions, which in most cases was accompanied by acidification [44-46]. Abnormal
153 (teratological) diatoms, which deviate from their usual morphological symmetry, have also been
154 used as bioindicators of exposure to toxic substances (usually metals) in both ecotoxicological
155 [47-48] and paleolimnological studies [49-50].

156 A few species of diatoms (e.g. *Navicula libonensis*) are used routinely to assess toxicity
157 of chemicals for use in risk assessments [51]. Benthic diatoms are common pollution indicators
158 in field assessments and mesocosm experiments to assess river pollution [52-53]. Various biotic
159 indices, including species diversity, autoecological metrics, and abundance of select indicator
160 taxa are used in diatom-based bioassessments of pollution [52]. These indices can also be applied
161 in diatom-based paleolimnological studies, although inferences of diversity from sediment cores
162 can be problematic, as changes in sedimentation rates that are unaccounted for can artificially
163 alter diversity inferences [54]. Still, these challenges can be overcome, and paleolimnological
164 approaches have much to offer the study of aquatic biodiversity [55], including the impacts of
165 toxic substances.

166 To date, most of the diatom-based paleolimnological assessments of pollution have
167 focused on acidification, eutrophication, and metals [49, 56-58], while an increasing number of
168 ecotoxicological studies also focus on diatom responses to persistent organic pollutants such as
169 pesticides and hydrocarbons [59-60]. For example, Bayona et al. [53] exposed diatom
170 communities in biofilms to thiram and a hydrocarbon emulsion in outdoor flow-through
171 mesocosms, and measured several structural and ecological traits to determine which measures
172 were most sensitive. They found that ecological guilds were more sensitive to these chemicals

173 than traditional bioindicator indices used for water quality assessments (e.g. for eutrophication).
174 For example, motile diatom species were more tolerant of persistent organic pollutants than non-
175 motile species [53, 59, 61]. Increased incidences of teratology have also been observed for
176 benthic diatoms exposed to the genotoxic herbicide, maleic hydrazide [62]. Paleolimnological
177 approaches have yet to be widely applied to the study of temporal changes in lake ecosystems
178 occurring in response to pesticides, hydrocarbons, or other persistent organic pollutants, with
179 some exceptions [63-64].

180

181 *Chironomids*

182 Non-biting midge larvae (Diptera, Chironomidae) are benthic macroinvertebrates that
183 leave subfossil remains in lake sediments in the form of chitinized head capsules that can often
184 be identified to the species-level [65-66]. Generally considered to be tolerant of pollution relative
185 to other macroinvertebrates, chironomid communities have been an integral component of field
186 and lab-based bioassessments of toxic substances [67-68], and several species in the *Chironomus*
187 genus are common model organisms in ecotoxicology [69]. In paleolimnology, chironomids are
188 commonly used in paleoclimate assessments [70], and are also used frequently to infer
189 eutrophication and reconstruct changes in deepwater oxygen concentrations [71-73].

190 Chironomids typically live associated with the sediments, and therefore direct
191 relationships between sediment quality (including concentrations of sediment-associated
192 contaminants like pesticides) and the abundance and species assemblages of chironomid
193 subfossil remains can be assessed through time in sediment cores. For example, chironomid
194 subfossil remains were examined in tandem with reconstructions of DDT and its metabolites in a
195 sediment core collected from a lake downstream of the St. Lawrence River at Montreal, Quebec

196 (Canada), where more than 16,000 kg of DDT were applied between 1965 and 1967, to
197 investigate downstream ecological impacts of pesticide application [63]. Concentrations of DDT
198 and its metabolites (Σ DDT) exceeded sediment quality guidelines. Subfossil diatom assemblages
199 showed little detectable response to Σ DDT, but chironomid taxa that live directly associated with
200 sediments where Σ DDT partitions exhibited notable changes in abundance and species
201 assemblage, as the absolute and relative abundances of *Paratanytarsus* and *Psectrocladius*
202 declined markedly while Tanypodinae and *Tanytarsus* more than doubled following the
203 application of DDT in the 1960s.

204 Similar to teratological diatoms, chironomid mouthpart deformities have been reported in
205 higher percentages when exposed to various metals and other toxic substances [74-75]. However,
206 deformities can also be natural occurrences [76]. A recent meta-analysis of studies on
207 chironomid mouthpart deformities emphasized the importance of understanding background
208 deformity rates in a community or population before asserting causal linkages to pollutant
209 exposure [77]. Since one of the main advantages of a paleolimnological approach is the
210 characterization of background conditions typically over decadal to centennial (and even
211 millennial) timescales, and chironomid mouthparts are well preserved in sediments,
212 paleolimnology may represent a promising tool for refining the use of chironomid mouthpart
213 deformities as an early warning bioindicator of pollutant exposure [74].

214

215 *Cladocera*

216 Cladocera (Crustacea, Branchiopoda) are the main group of zooplankton that leave
217 abundant and readily identifiable remains preserved in sediments cores, and provide several
218 opportunities for the development of paleo-ecotoxicology. Many laboratory and mesocosm

219 experiments have been conducted for *Daphnia* and other cladocerans to establish dose-response
220 relationships with a wide variety of metals and industrial chemicals [78-80]. In addition to an
221 ever-growing library of sensitivity to toxic substances, *Daphnia* also have a fully sequenced
222 genome and a well-understood life history and ecology, and are considered a keystone species in
223 lakes [81].

224 A primary challenge that has historically hindered the use of Cladocera in
225 paleolimnological studies is the differential preservation of taxa, and only a few groups can be
226 identified to the species-level [82, 83]. For *Daphnia*, only the post-abdominal claws and resting
227 stages (ephippia) typically preserve in lake sediments, although headshields can preserve in rare
228 cases [84-86]. Post-abdominal claws and ephippia cannot be identified to the species-level, but
229 instead are assigned into one of two (sometimes three) species complexes, each containing
230 several different species [87]. Different species within these complexes may display widely
231 different ecological tolerances. For example, *Daphnia* generally have higher calcium optima
232 compared to other zooplankton taxa, and declines in *Daphnia* have been observed in response to
233 lakewater calcium decline (a legacy of acidification) in North America [88]. However, there are
234 two species of *Daphnia* (*D. catawba* and *D. ambigua*) that are able to tolerate low ambient
235 calcium concentrations [10]. These taxa cannot be resolved from other taxa within the broader
236 species complexes based on morphology of their subfossil remains [87]. Instead, molecular
237 genetics are required to distinguish *Daphnia* to the species-level, and well-developed techniques
238 exist for extracting DNA from the resting egg bank [89-90]. Dormant resting eggs remain viable
239 for decades, and can be hatched from different sediment intervals to examine evolutionary
240 change in populations throughout different points in a lake's history in response to
241 environmental pressures [91-92]. This application for subfossil cladocerans has made many

242 important contributions to ecotoxicological research, and consequently is discussed in greater
243 detail below, in the section titled “Resurrection Ecology.”

244 Because shells and headshields do not typically preserve, *Daphnia* cyclomorphosis in
245 response to different predators cannot be tracked from sediments. This is unfortunate, as changes
246 in water chemistry can limit their ability to produce morphological defenses, increasing *Daphnia*
247 vulnerability to predation [93]. Thus, a common and important sub-lethal effect of *Daphnia*
248 exposure to pollutants cannot be documented from lake sediments. Similarly, *Ceriodaphnia*,
249 *Simocephalus*, *Scapholeberis*, and *Diaphanosoma* leave only a few small body parts preserved in
250 the sediments [83], and are rarely reported in paleolimnological studies. In contrast, bosminids
251 and chydorids are well preserved in sediments, with headshields, shells, ephippia, and
252 postabdomens and/or postabdominal claws readily identifiable [83, 94]. Similar to *Daphnia*,
253 *Bosmina* morphology is influenced by its dominant predators, and exposure to carbaryl has been
254 linked to inhibition of anti-predator morphological defenses against copepod predators [95].
255 Since bosminid remains are well preserved, changes in size structure through time can be tracked
256 from sediment cores [96], and carefully designed paleolimnological approaches have some
257 potential use for testing hypotheses about the effects of low-level pollutant exposure on
258 zooplankton predator-prey dynamics.

259 Paleolimnological approaches focusing on bosminids and chydorids can also be used to
260 test the effects of pollutant exposure on life history traits, for example pollutant effects on sexual
261 versus asexual reproduction in zooplankton [97-98]. Cladocerans and several species of rotifers,
262 reproduce parthenogenically, where females produce clonal females asexually throughout most
263 of the growing season, and switch to sexual reproduction during stressful conditions (e.g.
264 overcrowding, limited food availability). Because the carapaces of parthenogenic females can be

265 distinguished from ehippial females in bosminids and chydorids, the ratio of parthenogenic to
266 ehippial females can be tracked through time from sediment cores to investigate changes in
267 reproductive strategies in response to environmental stressors [99].

268 An important consideration for the use of cladoceran subfossils in ecotoxicological
269 applications is that, for most species, exposure to pollutants is in the water column. Therefore,
270 unlike benthic chironomids, where assemblage/morphologies can be more directly linked to
271 pollutant concentrations in the sediments, the concentration of pollutants in sediment intervals do
272 not necessarily reflect the historical exposure of cladocerans to pollutants in the water column.
273 For example, pesticides and other polychlorinated biphenyls are hydrophobic, and bind readily to
274 sediments. Several metal(loid)s, including arsenic, are redox sensitive, and can be remobilized
275 from the sediments back into the water column during periods of anoxia. Metals (e.g. cadmium,
276 chromium) and certain pesticides are known to bioaccumulate in *Daphnia ehippia*, and
277 measurements of contaminants in subfossil ehippial remains have potential use to track
278 historical exposure to contaminants in the water column [100-101]. To use this tool most
279 effectively, further study is needed to understand the biological and chemical processes by which
280 certain contaminants are sequestered in *Daphnia ehippia*. However, sediments contaminated by
281 pesticides and fire retardants have also been shown to have significant negative effects on the
282 hatching success of *Daphnia* from dormant eggs [102-103], and so the concentrations of
283 pollutants in sediments do have some direct toxicological relevance to Cladocera.

284 While cladocerans are the best represented group of zooplankton in the sediment record,
285 other taxa leave some identifiable fossil remains that have utility for a paleo-ecotoxicological
286 approach. In Lake Orta, Italy, the abundance and diversity of rotifer (*Brachionus*) resting eggs
287 was examined in a paleo-ecotoxicological application to assess the ecological effects of severe

288 acidification and copper pollution [104]. The authors noted a persistence of viable rotifer and
289 hatched egg cases in sediment intervals deposited during a period of severe acidification and
290 copper contamination, and suggested that the production of resting eggs may have contributed to
291 rotifer survival while other zooplankton taxa were extirpated. The above described study
292 demonstrates the considerable potential that exists to develop a paleo-ecotoxicological approach
293 to answer questions regarding the direct effects of sediment pollutants on zooplankton
294 recruitment from resting eggs, an important source of colonizers for zooplankton communities in
295 lakes.

296

297 *Other Bioindicators*

298 Other notable groups of biota that can be identified in sediment cores from their subfossil
299 remains are the chrysophytes (algae), *Chaoborus* (phantom midges, macroinvertebrate
300 predators), ostracods (crustaceans), and arcellaceans (testate amoeba). Although used less
301 frequently than diatoms, chironomids, and cladocerans, other groups have considerable potential
302 for integration into a developing paleo-ecotoxicological framework. For example, *Chaoborus* are
303 traditionally used to indicate fish presence/absence, as certain species of *Chaoborus* cannot
304 coexist with fish [105]. They were a crucial component in early studies on the limnological
305 impacts of acidification (see “Acid Rain” below). They are also important predators on *Daphnia*
306 and other cladocerans [93], and may be analyzed alongside cladoceran subfossil remains to infer
307 the impacts of pollutant exposure on predator-prey dynamics [106]. The ostracods represent
308 another useful bioindicator group for sediment toxicity testing [107-109], and could complement
309 chironomid-based paleo-ecotoxicological studies in benthic habitats. Similarly, arcellaceans
310 (testate amoeba) are benthic organisms that have been used as paleolimnological indicators of

311 pH, oxygen, eutrophication, road salt, and metal pollution [110-112]. Macrofossils of aquatic
312 macrophytes can also be identified in lake sediments [113], and have been used previously as
313 part of a multi-proxy paleolimnological study to demonstrate that widespread application of
314 tributyltin, an anti-fouling paint applied to boats, resulted in a collapse of aquatic macrophyte
315 communities [64].

316

317 **Case studies in paleo-ecotoxicology**

318 *Acid rain*

319 Applied paleolimnology as a discipline gained prominence during the so-called “acid rain
320 debates,” as it was recognized that the unique long-term perspective provided by lake sediment
321 cores held the answers to some of the most pressing questions of the time, which could not be
322 answered through any other means. Most notably, while it was evident that many waterbodies on
323 acid sensitive (poorly buffered) bedrock downwind of industrial centres were acidic, the
324 argument was made by mainly vested industrial interest groups that these lakes were naturally
325 acidic, since no pH measurements were available from the pre-industrial period. Diatom and
326 chrysophyte species have well-defined, specific pH tolerances and optima, and thus it was
327 possible to develop transfer functions using newly emerging, state-of-the-art statistical tools to
328 quantitatively reconstruct changes in lakewater pH through time [114]. Using this approach,
329 research groups from both Europe and North America were able to show that these lakes were
330 not naturally acidic, but instead had undergone acidification consistent with the timing of
331 industrial development [39, 115]. Another key contribution of paleolimnology to our
332 understanding of the ecological impacts of acidification was to show that, in many cases,
333 acidification led to the extirpation of fish [116]. Certain species of *Chaoborus* (e.g. *C.*

334 *americanus*) are heavily preyed upon by fish because they are large-bodied and do not undertake
335 diel vertical migration as a behavioural adaptation for predator avoidance [117]. As such, these
336 species do not typically co-exist with fish, and their presence in lake sediment cores is a strong
337 indicator of fishless conditions in North American waters [105]. Observations of the appearance
338 of *C. americanus* and other fishless indicators in dated sediment profiles, occurring coincident
339 with diatom-inferred changes in lakewater pH, provided strong evidence that lake acidification
340 can lead to the extirpation of fish. [116, 118-119].

341 Paleolimnological approaches continue to make important contributions to our
342 understanding of the processes of chemical and biological recovery following acidification, by
343 providing necessary information on pre-acidification conditions and the range of natural
344 variability [49, 120]. For example, Jeziorski et al. [88] used a paleolimnological approach,
345 interpreted in the context of lab bioassays and long-term monitoring [25], to demonstrate that
346 lakewater calcium decline, a legacy of decades of acid rain, negatively impacts *Daphnia*
347 populations and can inhibit recovery of zooplankton from acidification. Labaj et al. [121]
348 compared multi-proxy palaeolimnological records from Sudbury (Canada) and nearby Killarney
349 Provincial Park to tease apart the influence of metal contamination on biological recovery from
350 acidification. The authors inferred that biological recovery in Sudbury lakes is likely inhibited by
351 metals, which still exceed provincial water quality guidelines. These barriers to recovery are not
352 readily apparent when only examining short-term mesocosm or lab studies.

353 The integration of paleolimnology into an extensive, interdisciplinary network of
354 researchers studying acidification provides a useful roadmap for conceptualizing how
355 paleolimnological approaches can be more effectively synthesized into ecotoxicological research
356 moving forward. Mainly, key research questions were identified from the laboratory and field

357 ecotoxicological research (are acidic lakes natural, did acidification lead to the extirpation of
358 fish), and an innovative paleolimnological approach was devised (quantitative pH reconstruction,
359 *Chaoborus*-based inferences of fish presence/absence) to leverage the unique perspective
360 provided from the study of lake sediment cores to answer those questions.

361

362 *Resurrection Ecology*

363 *Daphnia* and several other species of zooplankton leave dormant resting stages in the
364 sediments. These resting stages can be isolated from dated sediment cores and hatched to
365 resurrect populations that existed at different time periods within a lake's history (termed
366 "resurrection ecology") [92, 122-123]. Resting eggs potentially as old as 700 years have been
367 hatched successfully in the lab [124]. Resurrection ecology provides the most direct example of
368 how paleolimnology can be integrated with classical toxicity testing. Resurrected zooplankton
369 populations from different time periods within a lake's history can be used in laboratory testing
370 to determine if they differ in terms of their tolerances to a stressor, and to establish the genotypic
371 and phenotypic basis for any observed differences [91, 124-125].

372 Zooplankton life history involves a sexual reproductive stage and short generation times,
373 and thus potential exists for populations to rapidly evolve tolerances to a contaminant or other
374 stressor. One of the fundamental questions that can be answered with resurrection ecology is if
375 *Daphnia* (or other zooplankton taxa) become more tolerate of a contaminant with long-term
376 exposure. If this is the case, then lab bioassays that are conducted on timescales of hours to
377 weeks may actually over-estimate the sensitivity of natural populations to a contaminant, where
378 sufficient time exists for evolutionary processes to ameliorate the harmful effects. This approach
379 has been used to investigate evolutionary responses of *Daphnia* and other zooplankton taxa to

380 acidification [126], metals [127-128], and pesticides [129-130]. For example, ancient genotypes
381 of *Daphnia pulicaria* have been reported to be more sensitive to chlorpyrifos (an
382 organophosphate insecticide) than younger genotypes, determined through acute toxicity tests
383 [130]. In contrast, Rogalski [131] observed the opposite: *Daphnia ambigua* clones hatched from
384 sediment intervals corresponding to a historical period of peak copper and cadmium
385 contamination were in fact more sensitive than ancient clones hatched from the pre-impact
386 period, apparent evidence of maladaptation to metal contamination. Future studies such as these,
387 which pair resurrection ecology with paleolimnological reconstructions of contaminant
388 deposition histories, can provide further insights into the trajectories of population evolutionary
389 responses to long-term contaminant exposure.

390 The viability of dormant resting eggs can decrease over time, which may be an important
391 confounding factor when interpreting differences in clonal tolerances to environmental
392 conditions hatched from different time periods [132]. Recently, Rogalski [128] conducted a
393 study to examine the combined influence of sediment age and metal contamination on both the
394 hatching rate of *Daphnia* diapausing eggs, and juvenile mortality of hatched *Daphnia* in four
395 Connecticut (USA) lakes over the past 40-100 years. The results showed that both sediment age
396 and metal contamination were negatively correlated to hatching rate of diapausing eggs, but that
397 metal contamination alone was positively, significantly associated with juvenile mortality of
398 hatched eggs. This study demonstrates that it is not only sediment age which may influence
399 hatching success of diapausing eggs, but also the environmental conditions in which the
400 diapausing eggs were produced. Rogalski [128] provides several interesting explanations for
401 possible mechanisms by which exposure to metals may influence hatching success of *Daphnia*,

402 including potential fitness costs of bioaccumulation of metals, maternal investment, and
403 genotoxic effects.

404 In summary, resurrection ecology provides unique insights into both evolutionary
405 responses to long-term exposure to pollutants, and the effects of contaminant exposure on the
406 functioning of sediment resting egg banks, both of which play a crucial role in long-term
407 zooplankton population and community dynamics.

408

409 *Recent studies on arsenic exposure*

410 The toxic effects of long-term exposure of aquatic biota to arsenic has received relatively
411 little attention (compared to copper, for example), despite being a prominent contaminant
412 globally. Within the last year, two independent studies [133-134] were published that each
413 examined the toxic effects of arsenic on multiple trophic levels under different scenarios and
414 geographic settings. Chen et al. [133] collected sediment cores from two lakes in southwest
415 China with documented histories of arsenic contamination from industrial tailings. Sediments
416 clearly recorded arsenic enrichment consistent with the known history of arsenic contamination.
417 Cladocera and diatoms were examined as ecological indicators of the potential toxic effects of
418 arsenic. Both daphniid and bosminid fluxes (an estimate of abundance) decreased abruptly with
419 arsenic contamination, a dramatic reversal of an increasing trend that was occurring with
420 ongoing eutrophication. *Achnantheidium minutissimum*, a benthic diatom taxon known to be
421 tolerant of metals, increased following arsenic contamination, while *Fragilaria contruens* and *F.*
422 *crotensis* decreased or were presumably extirpated. For both diatoms and cladocerans, critical
423 ecological transitions occurred consistent with the increase in arsenic concentrations in water
424 above 100 $\mu\text{g L}^{-1}$. No biological recovery was evident, despite recent decreases in arsenic.

425 Chen et al. [133] provided some of the first evidence of the long-term ecological
426 consequences of arsenic contamination in freshwater ecosystems at the community level, and
427 was rapidly followed by a second study also showing dramatic ecological impacts of arsenic
428 exposure. Thienpont et al. [134] examined diatoms, Cladocera, chironomids, and *Chaoborus* in a
429 naturally fishless, subarctic Canadian lake (Pocket Lake, Yellowknife, Northwest Territories)
430 historically impacted by gold mining activities that released toxic arsenic trioxide dust as a
431 byproduct. In contrast to the preceding example, Pocket Lake received arsenic contamination
432 solely from atmospheric sources, rather than tailings. At the height of mining, arsenic
433 concentrations in Pocket Lake were enriched by 1400%, and clear enrichment, albeit of a smaller
434 magnitude, was also evident for antimony, mercury, and lead. At the height of mining activities,
435 arsenic concentrations in the sediments were almost 4% arsenic by dry mass. Planktonic diatoms
436 were lost from the sediment record, while benthic diatom taxa like *A. minutissimum* increased,
437 consistent with Chen et al. [133]. In contrast, at the onset of arsenic pollution, *Daphnia* appeared
438 for the first time in the sediment record and increased in abundance until the height of pollution,
439 after which *Daphnia* and all cladocerans were functionally extirpated. *Daphnia* are known from
440 lab toxicity tests to be relatively tolerant of arsenic [135]. The findings from Pocket Lake support
441 this, suggesting that more sensitive, resident zooplankton populations were lost as a result of
442 arsenic contamination (perhaps copepods which do not preserve well in the sediments, and
443 receive less attention in ecotoxicological research; 136), leaving a niche available for *Daphnia* to
444 exploit. This suggests that *Daphnia* may not be the most sensitive model organism for assessing
445 the toxic effects of arsenic on waterbodies. However, arsenic contamination quickly exceeded
446 the tolerances of *Daphnia*, leading to their eventual extirpation [134]. No recovery was evident
447 for diatoms or Cladocera. Assemblage shifts in chironomids were observed consistent with the

448 history of arsenic pollution, but no taxa were extirpated and some recovery was evident. No
449 evidence of an impact of arsenic on *Chaoborus* was observed.

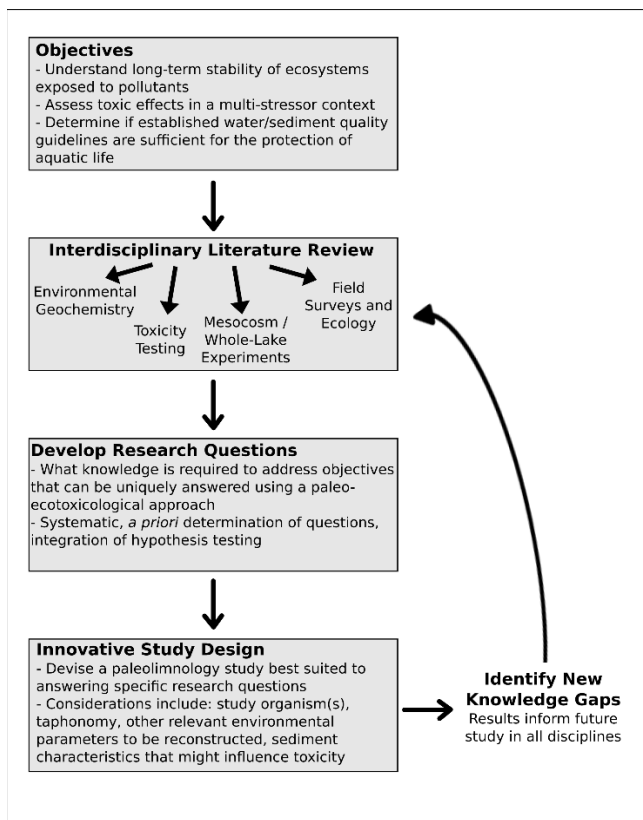
450 A third study [137], published within the same timeframe, contained similar themes to
451 those in Thienpont et al. [134] and Chen et al. [133], although arsenic was not the primary
452 contaminant of concern, but only one among many metal(loid)s (mainly Cd, Zn, and Cu). Ross
453 Lake in Flin Flon, Manitoba (Canada) received metal-contaminated sewage effluent, and like
454 Pocket Lake and the two lakes in Southwest China, an increase in metal(loid) concentrations was
455 followed by a period of recovery following improved tailings management. Metal concentrations
456 were consistent with the known pollution history, and the biodiversity of diatoms was reduced,
457 as was the abundance of Cladocera (particularly *Bosmina*) and *Chaoborus* [137]. Assemblage
458 changes in chironomids also occurred consistent with metal pollution. As was the case in the
459 previous two examples, minimal signs of recovery were evident despite decreases in metal
460 concentrations. An important finding of this study was that pre-industrial levels for several
461 metal(loid)s also exceeded Canadian Council of Ministers of the Environment (CCME) probable
462 effects guidelines for the protection of aquatic life, suggesting they are naturally elevated in this
463 system. The authors stressed the need for appropriate knowledge of baseline conditions to assess
464 toxic effects or remediation criteria. All three of these examples demonstrate how the dramatic
465 ecological impacts of pollutants are clearly visible when long timescales are considered, and
466 noise from inter-annual variability is reduced.

467

468 **Recommendations for a new paleo-ecotoxicological framework**

469 Paleolimnology has made significant contributions to our understanding of the movement
470 of contaminants in the environment, and their impacts on lake ecosystems. With regards to the

471 latter, however, our view is that the potential of paleolimnology to answer key questions related
 472 to toxic effects on aquatic biota has not been realized, due, in part, to limited communication
 473 amongst paleolimnologists and ecotoxicologists. Here, we provide recommendations for a new
 474 paleo-ecotoxicological framework (Figure 2), with the aim of stimulating development of new
 475 paleo-ecotoxicological approaches that use a long-term perspective to answer pressing questions
 476 in ecotoxicology, moving beyond simply providing an assessment of natural variability and
 477 baseline conditions, valuable though this information is.



478

479 **Figure 2.** Our recommendation for what a new paleo-ecotoxicological framework could look like. Central
 480 objectives related to long-term ecological impacts of pollutant exposure in a multi-stressor context and evaluating
 481 regulatory guidelines are well suited to a paleoecotoxicological approach. A critical review of interdisciplinary
 482 literature can identify important knowledge gaps that can be addressed effectively using lake sediment records. A
 483 systematic study design, targeted to answering clearly defined a priori research questions, can take advantage of the
 484 wealth of information preserved in lake sediments. In turn, results of paleo-ecotoxicological studies can identify
 485 new knowledge gaps that can be addressed through studies using multiple approaches (e.g., toxicity testing,
 486 geochemistry). In this way, paleolimnology can be seamlessly integrated into an interdisciplinary context to provide
 487 novel insights into the toxic effects of pollutants on aquatic ecosystems.

488 A paleolimnological approach is well suited for investigations into the long-term stability
489 of ecosystems exposed to toxicants, with well-developed techniques available for inferring both
490 historical trends in key contaminants of interest, and ecological changes in key groups of aquatic
491 biota. Importantly, it is feasible to analyze lake sediment cores from a suite of lakes in a region
492 of interest, providing opportunities to compare ecotoxicological effects in lakes subjected to
493 different environmental stressors, across wide limnological gradients, and with different pre-
494 disturbance ecological communities. This, in combination with a long-term perspective over
495 centennial to millennial timescales, also makes a paleolimnological approach well-suited for
496 assessing toxic effects in a multi-stressor context across a full range of environmental conditions.
497 Quasi-experimental designs that analyze sediment cores from strategically selected lakes [e.g.
498 34, 138] can be set up to further aid in the disentangling of multiple stressors, and influences of
499 different water chemistry conditions (e.g. alkalinity) on ecosystem toxicity. Ultimately, a central
500 objective of an emerging paleo-ecotoxicological framework should be to provide a means to
501 evaluate the applicability and effectiveness of sediment and water quality guidelines for the
502 protection of aquatic life. These general objectives can be applied across broad geographical
503 scales, to address the ecological impacts of many potential contaminants, or to assess the impacts
504 of industrial activities on specific regions of interest.

505 Paleolimnology is interdisciplinary by nature, and a paleo-ecotoxicological study requires
506 a strong grasp of the available knowledge on environmental geochemistry (how a chemical of
507 interest behaves in the environment), the mode of toxic action of that chemical, and its known
508 ecological effects across different spatial and temporal scales (i.e. integrating lab toxicity testing,
509 field surveys, ecosystem experiments). A thoughtful evaluation of the literature, including a
510 critical review of the strengths and limitations of different study designs, is useful for identifying

511 where critical knowledge gaps exist that can be addressed using a paleo-ecotoxicological
512 approach. Systematic study design, with clearly defined *a priori* research questions
513 (incorporating hypothesis testing where appropriate), should be developed with the overlying
514 objectives in mind. An adaptable paleolimnological study can then be designed to most
515 effectively answer the *a priori* established questions. For example, which biological proxies are
516 best suited to address the questions of interest? Is a multi-indicator assessment most appropriate
517 [139], or should the focus be on evaluating specific organisms (i.e. *Daphnia*) used in lab toxicity
518 testing in a real-ecosystem setting? Is the main route of toxic exposure the sediments or in the
519 water column (i.e. are we evaluating the efficacy of sediment or water quality guidelines), and
520 which indicator(s) will best facilitate evaluation of this exposure route? Paleolimnology is a
521 rapidly growing discipline, and as the ability to reconstruct past environmental conditions using
522 lake sediment records continues to improve with time, so will the ability to ask increasingly
523 nuanced questions about the toxic effects of pollutants. In particular, rapid advances in
524 environmental DNA (eDNA) and DNA sequencing technology represents a powerful tool for
525 detecting organisms that do not leave identifiable subfossil remains in sediment cores [140]. An
526 interesting example of this emerging paleo-ecotoxicological tool can be found in Poulain et al.
527 [141], where DNA extracted from a dated sediment core revealed an association between the
528 phylogeny of the microbial mercuric reductase gene (*merA*) and the onset of the industrial
529 revolution, evidence of a rapid evolutionary response by microbial communities to
530 anthropogenic mercury deposition.

531 There are many opportunities available to directly apply tools used in ecotoxicology and
532 environmental chemistry to evaluate the bioavailability and toxicity of the sediment matrix
533 through time using lake sediment cores. For example, Xiao and colleagues [142] characterized

534 the sediment matrix at varying intervals in a sediment core, including organic carbon content,
535 grain size, and the chemical speciation of metals, and related this information to sediment risk
536 assessments in order to evaluate changes in ecological risk through time in Lianhuan Lake,
537 eastern China. This approach could be further expanded by examining the subfossil remains of
538 benthic organisms that live associated with the sediments, in the same sediment intervals, in
539 order to directly infer impacts on biotic assemblages. For example, we can characterize changes
540 in the chemical make-up of sediment intervals and directly relate this to chironomid abundances
541 and species assemblage, or the prevalence of chironomid mouthpart deformities, to determine at
542 what concentration toxic effects are observable. It is more challenging to link sediment core
543 proxies to historic changes in contaminant exposure in the water column, as the concentrations of
544 chemicals measured in the sediments does not necessarily reflect what organisms were exposed
545 to. It is possible, however, to provide indirect information on historic contaminant exposure in
546 the water column. The potential also exists to use knowledge of chemical kinetics to estimate
547 water column concentrations from sediment values (e.g. fugacity modeling), however this is not
548 a straightforward process. This does not negate the effectiveness of a paleo-ecotoxicological
549 approach, but does highlight the importance of an understanding of biogeochemistry and
550 taphonomy (the process of fossilization/preservation in sediments) when designing and drawing
551 conclusions from paleo-ecotoxicological studies. In cases where the sediment core record can be
552 compared against limnological monitoring records (e.g. Chen et al [133]), it may still be possible
553 to use the sediment core record to establish what concentrations of a pollutant cause ecological
554 effects. Some pollutants can also be measured directly in subfossil remains of pelagic organisms
555 (e.g. *Daphnia ephippia* [100]) to reconstruct historical exposure, though an understanding of
556 contaminant partitioning and the related physiological processes that influence the concentration

557 of pollutants in subfossil remains is beneficial for proper interpretation. The geochemical
558 characterization of subfossil remains could also be examined for other relevant ecotoxicological
559 questions, such as a record of changes in carbon cycling and food web structure (e.g. carbon and
560 nitrogen stable isotopes) [143-144], in addition to providing a record of exposure to pollutants, to
561 provide further insights into the chronic effects of pollutants on aquatic ecosystems.

562 Paleo-ecotoxicology as a sub-discipline has begun to gain prominence, especially within
563 the last several years [31, 104, 127, 134 137]. It is increasingly recognized that lake sediment
564 cores can provide novel information on the responses of aquatic ecosystems to many different
565 pollutants, and is highly complementary to other ecotoxicological approaches. Like whole-
566 ecosystem experiments, paleo-ecotoxicology allows for investigations into ecosystem-wide
567 responses to pollutants, although mechanistic links are more challenging to tease out, and there is
568 less temporal resolution, but longer timescales can be examined. However, paleolimnological
569 studies can be replicated more easily than expensive and logistically-intensive whole-ecosystem
570 experiments, both within a lake and on a regional scale, to span relevant limnological gradients.
571 As part of an integrated framework, with better interdisciplinary collaboration and integration
572 within ecotoxicology, paleo-ecotoxicology can leverage strengths and limitations across
573 disciplines. Toxicity testing (lab and mesocosms), reductionist by design, provide the
574 mechanistic, cause-effect basis for designing and interpreting biotic changes reported in paleo-
575 ecotoxicological studies. Mesocosms, field surveys, whole-ecosystem experiments, and other
576 ecological approaches can be used to infer the role of indirect, ecological effects of toxic
577 exposures. With a solid grounding in the breadth of interdisciplinary literature available, the
578 paleo-ecotoxicologist can develop targeted study designs that address key knowledge gaps,

579 contributing to a more holistic picture of the toxic effects of pollutants. Knowledge which is
580 necessary for establishing effective regulatory guidelines to protect aquatic life.

581

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587

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