Environmental Impact Statement

Few studies are available for understanding the seasonal distributions of PAHs and OCPs in water from the well-known hyper-eutrophic Lake Taihu. These hydrophobic contaminants could be easily further accumulated in phytoplankton, and then will either be grazed on and transported to other compartments of the pelagic food chain or carried to the sediment by live or dead cells and sinking aggregates. It is necessary to investigate the potential ecological effects of phytoplankton on the fate of hydrophobic organic compounds (HOCs), which will be helpful for elucidating the interactions between HOCs and eutrophication from a holistic approach.

1   **The potential effects of phytoplankton on the occurrence of organochlorine**

2   **pesticides (OCPs) and polycyclic aromatic hydrocarbons (PAHs) in water from**

3   **Lake Taihu, China**

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5   **Abstract**

6   Seasonal distribution of organochlorine pesticides (OCPs) and polycyclic aromatic

7   hydrocarbons (PAHs) and possible interactions with phytoplankton species in water

8   from Lake Taihu, were investigated. OCP concentrations ranged from 69.95 to 223.08

9   ng L^{-1} in winter and from 80.95 to 376.03 ng L^{-1} in summer, while PAHs ranged from

10   45.40 to 232.74 ng L^{-1} in winter and 49.53 to 197.72 ng L^{-1} in summer. Such

11   physicochemical and biological parameters as the larger amounts of pollutants

12   uptaken by phytoplankton, and the increased atmospheric wet deposition, the

13   discharge of wastewater, the resuspension of polluted sediments in summer time, were

14   responsible for the relative higher residues of both OCPs and PAHs than winter.

15   Canonical correspondence analysis (CCA) between phytoplankton biomass and

16   micropollutants indicated high affinity of OCPs to *Bacillariophyta* and *Cryptophyta*

17   and PAHs to *Cyanophyta* and *Chlorophyta*, documenting the ecological effects of

18   phytoplankton on the biogeochemical processes of OCPs and PAHs and thus should

19   be further investigated especially in hyper-eutrophic lakes.

20   **Introduction**

21   National and international pressure is mounting for water quality improvement in the

22   twenty first century, and it is commonly known that a significant threat to water

23   quality is posed by the discharge and dumping of wastes containing high levels of

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24   anthropogenic organic micropollutants such as polycyclic aromatic hydrocarbons

25   (PAHs) and organochlorine pesticides (OCPs).[1] PAHs and OCPs are typical toxic

26   organic pollutants and most are listed as priority pollutants by national organizations

27   due to their persistence, long-range transport, bioaccumulation and biological

28   toxicity.[2,3] They can enter the aquatic environment through different pathways,

29   including effluent discharge, agricultural runoff, atmospheric deposition, and air/water

30   exchange.[4] Therefore, PAHs and OCPs have been widely detected and become

31   ubiquitous pollutants in freshwater and marine aquatic ecosystems, and have received

32   considerable worldwide attention resulting from public awareness of environmental

33   problems and expectations for a good quality of life.[5,6]

34   During the last decades, increased primary production has been observed in many

35   lakes and most coastal areas from enhanced anthropogenic inputs of nutrients.

36   Eutrophication in freshwater and coastal marine areas includes a series of processes

37   that alter the ecosystems significantly.[7] An increased nutrient input causes a stepwise

38   change in productivity, with phytoplankton biomass increase being one of the most

39   obvious effects. Furthermore, increasing biomass and phytoplankton growth rates will

40   finally influence the biogeochemical cycles of hydrophobic organic compounds

41   (HOCs), such as OCPs and PAHs, in aquatic environments due to the uptake of these

42   pollutants by phytoplankton.[8,9] And then, contaminants accumulated in phytoplankton

43   will either be grazed on and transported to other compartments of the pelagic food

44   chain or carried to the sediment by live or dead cells and sinking aggregates.[10] It is

45   important to elucidate the effects of eutrophication on HOC uptake in phytoplankton,

46   and whether these effects are transferred to, and sustained through, the aquatic food

47   webs. Few investigations have, however, addressed this issue. Traditionally, the

48   environmental effects of eutrophication and contaminants have been studied

49 separately without a holistic approach.[9] In recent years, potential interactions of

50 eutrophication and contaminant cycling have been highlighted among other things,

51 resulted in the Eutrophication and CONtaminants (EUCON) research program of the

52 Swedish Environmental Protection Agency.[11] Although numerous investigations have

53 been conducted over the last decades on bioconcentration of HOCs by phytoplankton,

54 the influence of the biomass and community compositions of phytoplankton on the

55 fate of HOCs still remain to be clarified.[12]

56 The Yangtze River Delta is one of the most important economic engines of China,

57 characterized by fast urbanization and industrialization.[13] The Lake Taihu basin, the

58 heart of the Yangtze River Delta, is also an important grain production base in China

59 and as such has the highest pesticide application in the country. The basin covers only

60 0.4% of the total territory of the country but contributes 3.1% of the national food

61 production and 10% of the gross domestic product. With rapid economic development

62 and population increases, man-made pollutants derived from household, agricultural,

63 and industrial activities have been discharged into the water, resulting in serious

64 pollution of HOCs and eutrophication in Lake Taihu.[14] However, most previous

65 studies focused on the residual levels and possible ecological risk of PAHs and OCPs

66 in different environmental compartments including air, water, sediment, benthos,

67 fish.[15-20] Knowledge of the seasonal distribution of such pollutants in the water

68 column is lacking. Moreover, Lake Taihu has also experienced accelerated

69 eutrophication over the past 3 decades and has changed from a mesotrophic,

70 diatom-dominated lake to a hyper-eutrophic, *Cyanophyta*-dominated system, with

71 *Microcystis* blooms now occurring regularly throughout much of the lake.[21] Since

72 most environmental decisions around this area are based on knowledge of separate

73 effects of either HOCs or eutrophication, ignoring interactive processes that may

74 influence the overall effects, knowledge about potential interactive mechanisms is

75 desirable both with respect to remedial action and as a basis for the development of

76 environmental criteria and classification of contaminants with respect to risks.

77   The objective of the present study was to investigate the spatial and seasonal

78 distribution of PAHs and OCPs in water samples from Lake Taihu, a hyper-eutrophic

79 lake in China, and thus enhancing the public awareness of toxic organic pollutants in

80 freshwater lakes. Moreover, we investigated the phytoplankton community structure

81 and their possible effects on the fate of such hydrophobic chemicals to determine

82 potential interactions between organic pollutants and eutrophication in lake

83 ecosystem.

84 **Materials and methods**

85 **Sampling**

86 The detailed information of 15 sampling locations distributed over the whole lake are

87 shown in Fig. 1, included Meiliang Bay (TH1, TH2, TH3), Zhushan Bay (TH8), Lake

88 Center (TH4, TH9, TH10, TH12), Dapu Estuary (TH5), Jiapu Estuary (TH11), and

89 Xiaomei Estuary (TH6), South Lake Taihu (TH13, TH14, TH15) and East Lake Taihu

90 (TH7). Water samples for OCPs and PAH analysis were collected in January and

91 August 2010 corresponding to winter and summer samples, respectively. Surface

92 water samples at 0.5 cm depth were collected from each location using pre-cleaned 1L

93 brown glass bottles. The bottles were rinsed three times with samples and then

94 carefully filled just to overflowing without passing air bubbles through the samples.

95 All water samples were filtered through GF/C fiber glass filters (2.0 μm pore size,

96 Whatman, Britain) in the laboratory to remove sand and debris, and then stored in the

97 refrigerator at 4 °C prior to HOC extraction. The extraction of HOCs was completed

98　　within 2 days.

99　　　Samples for the phytoplankton analysis were conducted according to Chen et al.[22,23]

100　Integrated water samples were taken using a 2 m long and 10 cm diameter plastic tube.

101　Phytoplankton were then enumerated microscopically from samples fixed in Logul's

102　iodine and sedimented for 48 h. Biovolumes (ind L^{-1}) were calculated from cell

103　numbers and cell size measurements. These biovolumes were converted to biomass

104　based on the assumption that 1 mm^3 of volume equals 1 mg of fresh-weight biomass

105　(mg L^{-1}). Proportions of *Cyanophyta* (%) to total phytoplankton were calculated using

106　biovolume values.

107　　　While for the chemical properties of the water samples, the suspended solids (SS,

108　mg L^{-1}) were obtained by filtering water samples (100-500 mL according to the

109　amount of particles) using GF/F (0.7 μm pore size, Whatman, Britain) fiberglass

110　filters which had been pre-combusted at 550 °C for four hours. The filters were

111　subsequently dried at 105 °C and weighed using an electrobalance with an accuracy

112　of 0.01 mg. The filtration (20 mL) was then used for the determination of dissolved

113　organic carbon (DOC, mg L^{-1}) by catalytic combustion technique, which was

114　completed by the Torch organic carbon analyzer (Torch, Teledyne Tekmar, USA, the

115　relative standard deviation (RSD) <1.5%).

116　**Fig. 1**.

117　**Chemicals**

118　All chemicals and solvents used for OCP and PAH determination were pesticide

119　grade and purchased from Supelco (USA). The purified water was obtained from a

120　Milli-Q water system (Millipore, Bedford, MA, USA). OCP and PAH standards were

121　also purchased from Supelco. PAH standard solution contains 16 priority PAHs,

5

122   including naphthalene (NaP), acenaphthylene (Any), acenaphthene (Ana), fluorene

123   (Flu), phenanthrene (Phe), anthracene (Ant), fluoranthene (Flt), pyrene (Pyr),

124   benz[*a*]anthracene   (BaA),   chrysene   (Chr),   benzo[*b*]fluoranthene   (BbF),

125   benzo[*k*]flouranthene (BkF), benzo[*a*]pyrene (BaP), dibenz[*ah*]anthracene (DahA),

126   benzo[*ghi*]perylene (BghiP), and indeno[1,2,3-*cd*]pyrene (InP). OCP standard solution

127   contains α-, β-, γ-, and δ-HCH, heptachlor, heptachlor epoxide, α- and γ-chlordane,

128   aldrin, endrin, dieldrin, endosulfan I, endosulfan II, endrin aldehyde, endosulfan

129   sulfate, endrin ketone, p,p'-DDE, p,p'-DDD, p,p'-DDT, and methoxychlor. Both

130   stock   standard   solutions   were   dissolved   in   suggested   solvents   with

131   methanol/dichloromethane (1:1, v/v) for PAHs and n-hexane for OCPs, respectively.

132   The working solutions were stored at 4 °C and daily prepared as needed with suitable

133   dilutions before use.

134   **HOC extraction**

135   The filtered samples were extracted by solid phase extraction system equipped with

136   C18-boned phase columns (500 mg/6 mL of reversed phase octadecyl) purchased

137   from Supelco (USA). The column was conditioned with 20 mL methane followed by

138   20 mL deionized water at the rate of 5 mL min^{-1}. The filtered water samples (1 L)

139   were then injected to the cartridge, and the absorbed compounds were eluted with 10

140   mL methane and 20 mL dichloromethane. The elutions were combined and dried

141   through sodium sulfate (baked at 400 °C for 5 h) and then were evaporated to nearly 1

142   mL using a rotary evaporator (BUCHI 250, Sweden). Next, n-hexane was added to

143   redissolve the extractions and transferred to a 5mL amber vial for further purification

144   by a silica-alumina powder column (2:1) according to Zhao et al.[24] The target elutions

145   were collected and evaporated to near dryness with n-hexane as the substitute solvent

146   for the following instrumental analysis.

147   **Quantitative determination of HOCs**

148   The purified sample extracts for OCP analysis were quantitatively analyzed using an

149   Agilent 7890 gas chromatograph equipped with a [63]Ni μ-electron capture detector

150   (GC-μECD) and a HP-5MS fused silica capillary column (30 m × 0.25 mm × 0.25 μm,

151   J&W Scientific Inc., USA). PAHs were determined by high performance liquid

152   chromatography (HPLC) equipped with a diode-array detector (DAD) and a

153   series-wound fluorescene detector (FLD). Detailed instrumental conditions can be

154   found in Zhao et al.[25,26]

155   Identification of PAHs and OCPs was based on the retention time of standard

156   components analyzed under the same instrument conditions, while the quantification

157   was completed by the external standard method. The method detection limits of

158   individual compounds using the present methods were determined as the

159   concentration of analytes in a sample that yield a peak signal-to-noise ratio (S/N) of 3,

160   in the range of 0.01-3.52 ng L^{-1} for PAHs and 0.01-2.57 ng L^{-1} for OCPs, respectively.

161   For each set of samples, a procedure blank and a spiked sample consisting of all

162   reagents were run to check for interference and cross contamination. Results of

163   laboratory blanks indicated no interferent contaminants were detected during the tests,

164   and the spiked recoveries performed with deionized water ranged from 75% to 105%

165   for PAHs and from 78% to 112% for OCPs, respectively. Relative standard deviations

166   conducted with standard solutions were all less than 5% for both compounds.

167   Additionally, standard solution was added every 15 samples to recalibrate the

168   retention time of target compounds.

169　**Statistical analysis**

170　Statistical treatment of the obtained results and related correlation analysis were

171　performed with SPSS software (SPSS 16.0 for Windows, USA). Spearman's rank

172　correlation was used to test the strength of associations between parameters, and the

173　statistical significance was considered for $p<0.05$ (two-tailed tests). Canonical

174　correspondence analysis (CCA, CANOCO 4.5 for Windows, USA), a simple method

175　for arranging biological species along environmental variables, was also used to

176　examine the features of the distributions of species along the environmental variables

177　to determine the maximum correlations between pollutant concentrations and the

178　biomass of phytoplankton species.

179　**Results and discussion**

180　**Concentrations and distribution of OCPs and PAHs**

181　The concentrations and distribution of OCPs and PAHs in water samples collected in

182　winter and summer from Lake Taihu are shown in Fig. 2. In winter, the concentrations

183　of OCPs ranged from 69.95 to 223.08 ng L^{-1}. The highest concentration was found at

184　Meiliang Bay (TH1), followed by the Lake Center area (TH10: 207.56 ng L^{-1}) and

185　Dapu Estuary (TH5: 185.70 ng L^{-1}), while the lowest concentration was detected at

186　East Lake Taihu (TH7). Low OCP residues with 87.89 ng L^{-1} concentration were also

187　observed at the Lake Center area (TH4). Compared to the higher concentrations of

188　OCPs detected in parts of the estuary regions (Dapu Estuary TH5 and Jiapu Estuary

189　TH11: 160.17 ng L^{-1}), OCPs were found at a lower concentration of 98.42 ng L^{-1} at

190　Xiaomei Estuary, which might be attributed to the different input burdens around this

191　area.[15] In summer, OCPs were found ranging from 80.95 to 376.03 ng L^{-1}, and the

192　highest and lowest concentrations were detected at the Jiapu Estuary (TH11) and

193   South Lake Taihu (TH15), respectively. Higher residues were also found at Meiliang

194   Bay (TH1), Centre Lake (TH4), and Xiaomei Estuary (TH6) than other sampling sites,

195   with the observed values of 321.89, 259.58 and 253.75 ng L^{-1}, respectively. Compared

196   with the winter samples, higher values were found in summer except for sites TH5,

197   TH10, TH11, and TH12. Additionally, variance analysis performed by independent

198   samples T-test indicated significant variance was observed between the two seasons

199   based on the concentrations of total OCPs (p=0.039<0.05, n=15), thus demonstrating

200   the seasonal difference of OCPs in water column from Lake Taihu.

201   As for PAHs, they were found with concentrations ranging from 45.40 to 232.74 ng

202   L^{-1} and the coefficient of variance among different sites was 0.57 in winter. The Jiapu

203   Estuary (TH11) had the highest residual level, followed by the Centre Lake area

204   (TH10: 231.77 ng L^{-1}) and South Lake Taihu (TH13: 225.92 ng L^{-1}), while TH4 had

205   the lowest value. The Meiliang Bay, including sites TH1, TH2, and TH3, also had

206   high concentrations of 137.56, 68.20, and 129.48 ng L^{-1}, respectively, which may be

207   related to the extensive discharge of pollutants around these areas. Industries in the

208   area surrounding Meiliang Bay are primarily composed of machine shops, paper mills,

209   textile mill, chemical plant, leather industries power plants, and other manufactures,

210   all of which contributed to the relative high PAH residues in Meiliang Bay.[20] PAHs in

211   the estuaries such as Dapu Estuary (TH5) and Xiaomei Estuary (TH6) were 69.76 and

212   131.02 ng L^{-1}, respectively, possibly indicating a heavy discharge of wastewater from

213   tributaries, which was consistent with a significant positive relationship between

214   drainage area and the level of PAHs suggested by Weinstein et al.[27] In summer, PAHs

215   ranged from 49.53 to 197.72 ng L^{-1} with the highest concentration at Xiaomei Estuary

216   (TH6), followed by Meiliang Bay TH1 (197.39 ng L^{-1}) and TH3 (193.43 ng L^{-1}), and

217   the lowest concentration was observed at South Lake Taihu (TH13). A high PAH

218 residue of 191.56 ng L^{-1} was also found in Zhushan Bay, one of the most polluted

219 areas in Lake Taihu due to the two severely polluted rivers (Taige Canal and Caoqiao

220 River) carrying contaminants directly into the lake.[16] Similar seasonal distribution of

221 higher residues of both OCPs and PAHs were observed in winter and summer for

222 several sites (TH10, TH11, and TH12), while PAHs found at other sites were higher in

223 summer than winter, although no significant difference was detected (p=0.738>0.05,

224 n=15).

225 **Fig. 2**.

226 **Seasonal variations of OCPs and PAHs**

227 As for the seasonal differences, most samples showed higher residues of PAHs in

228 summer than in winter time except for several sites (TH10, TH11, TH12, TH13) as

229 suggested by the abovementioned analysis. OCPs were also found the similar seasonal

230 variation with 73.3% of samples had higher concentrations than in summer.

231 Atmospheric deposition is considered to be an important input of PAHs to surface

232 waters; 10-80% of PAH inputs to the world's oceans are estimated to originate from

233 atmospheric sources.[28] Total PAH atmospheric inputs to the Mediterranean Sea are

234 estimated to be 3570 t yr^{-1}, with a mean value of 47.5 t yr^{-1}, whereas the total PAH

235 riverine inputs are about 5.3 and 1.3 t yr^{-1} from the Phone and Ebro rivers,

236 respectively.[29] Among atmospheric deposition, inputs of such pollutants to lakes due

237 to rainwater on numerous occasions have been shown to contain many organic

238 compounds including PAHs, and the concentration of PAHs in rainfall can sometimes

239 be much higher than in the receiving waterbody.[30] The dry and wet particle deposition

240 of dichlorodiphenyltrichloroethanes (DDTs) was more important than deposition

241 through the air-water exchange in the Great Lakes;[15] therefore, seasonal variations of

242 micropollutants were consistent with the seasonal distributions of rainfall with higher

243 rates in summer but lower rates in winter.

244   Furthermore, seasonal variation of PAHs and OCPs could be attributed to the

245 increased discharge of wastewater through tributaries during summer. The Taihu

246 Region is highly industrialized and surrounded by many industrial factories,

247 agricultural fields, livestock and aquaculture farming, and high-density populations,

248 which may discharge more contaminants directly into the lake,[20] especially in summer

249 due to the increased water flow from surrounding tributaries. In addition, as a typical

250 large shallow lake, resuspension of sediments induced by the heavier wind-wave

251 disturbance would also change the residues of pollutants in the aquatic phase.[31] SS in

252 water were higher in winter samples than in summer samples (Table 1), especially for

253 the sites at Centre Lake (TH4, and TH10), the estuary areas (TH5, TH6, TH11), and

254 the southern areas (TH13, TH14, and TH15) dominated by grass. The higher winter

255 values were attributed to the stronger wind velocity in winter.[32] We only noted higher

256 residues of PAHs winter than in summer at sites TH10, TH11, TH12, and TH13, and

257 higher winter OCPs at sites TH10, TH11, and TH12. This finding was consistent with

258 the seasonal variations of SS, thus demonstrating the limited influence of

259 micropollutant inputs to the water column by sediment resuspension compared to

260 atmospheric deposition and external discharge.

261   In addition, phytoplankton uptake coupled with air-water diffusive exchange will

262 also have an effect on input of HOCs to the aquatic system, which thus finally

263 determines their concentrations in water.[10,33] The transport of HOCs, apart from

264 catchment area input, across the surface microlayer is driven by atmospheric

265 deposition and counteracted by water-to-air volatilization.[15] This has also been

266 demonstrated by Qiu et al.[15] that atmospheric transport and deposition of endosulfan I

267   was a source of this compound to Lake Taihu. Furthermore, the link between

268   phytoplankton bioaccumulation and air-water exchange has been demonstrated by a

269   number of observations.[34,35] Dachs et al.[36] provided a first theoretical assessment of

270   the possible control of plankton biomass dynamics on both the water-dissolved phase

271   and air-water exchange of hydrophobic chemicals. Polychlorinated biphenyls (PCBs)

272   and PAH concentration profiles observed in the water column, either in the dissolved

273   phase or absorbed to phytoplankton, were similar to those found in the gas phase,

274   which suggested that air-water exchange supported and might even control the PCB

275   concentrations in phytoplankton.[37] As HOCs in the dissolved state were accumulated

276   by phytoplankton and later sedimented in association with sinking particles

277   accompanied by the crash of the plankton bloom, leading to a depletion of

278   dissolved-phase HOC concentration in surface water, promoting net diffusive transfer

279   from the atmosphere to the surface water.[33,38] As a result, gaseous transport of

280   semivolatile HOCs was higher in eutrophic systems such as Lake Taihu especially in

281   summer due to the higher amount of phytoplankton (Table 1).

282   **Table 1**.

283   **Species-specific affinity of phytoplankton to HOCs**

284   A complex succession of different planktonic species will have different ecological

285   implications of the fate of HOCs.[12] CCA is a nonlinear multivariate direct gradient

286   analytical method, which combines correspondence analysis with multiple regressions

287   and can provide detailed analysis of the relationships between biological species and

288   the environment.[39] Researchers have recently begun to use CCA to examine the

289   relationships between phytoplankton and water nutrient contents.[40] However, few

290   studies have used this method to examine the ecological contributions of

291   phytoplankton to bioaccumulation of toxic organic pollutants.[41] CCA was then used

292   here to determine the potential effects of phytoplankton species on the fate of

293   micropollutants in the water column of Lake Taihu. First, detrended correspondence

294   analysis (DCA) of concentrations of both PAHs and OCPs demonstrated that the first

295   axis length of the gradient was <2.0, and then the correlation analysis between the

296   concentrations of micropollutants and phytoplankton biomass was completed by the

297   redundancy analysis (RDA). The 2, 3, 4, and 5 ring PAH congeners were all

298   distributed at the second quadrant (Fig. 3), showing positive correlations with

299   *Aphanizomenon*, *Schroederia*, and *Ankistrodesmus* and relative weak positive

300   correlations with *Microcystis* and *Scenedesmus*, which demonstrated that *Cyanophyta*

301   and *Chlorophyta* were the major phytoplankton species affecting PAH distributions in

302   the water column. For OCPs, the major components including chlordanes, aldrins, and

303   DDTs were distributed at the first quadrant, in accordance with the fate of *Aulacoseira*

304   and *Asterionella*, and HCHs were positively correlated with *Cryptomonas*. Therefore,

305   OCPs in water were mainly correlated with *Bacillariophyta* and *Cryptophyta*, while

306   PAH residues in water showed positive correlations with *Cyanophyta* and

307   *Chlorophyta*.

308   **Fig. 3.**

309   These relationships indicated the potential influence of phytoplankton on the

310   biogeochemical cycling of micropollutants in aquatic ecosystems attributed to the

311   species-specific accumulation of HOCs by different phytoplankton.[10,12,42]

312   Phytoplankton uptake is a two-compartment process, with fast adsorption to the

313   phytoplankton cell surface followed by diffusion into the matrix in a partitioning-like

314   mechanism; therefore, the initial adsorption in the uptake of HOCs in phytoplankton

315   can be affected by the chemical composition of the cell wall.[42,43] The muramine acid

13

316 of *Cyanophyta* causes a lower uptake of HOCs than the phospholipid cell wall of

317 eucaryotic phytoplankton, such as *Bacillariophyta*.[10] Additionally, the lipid quality

318 will also influence HOC uptake in phytoplankton. PCBs are generally deposited in the

319 lipid stores of organisms, and Berglund et al.[12] found that lipid contents explained

320 most of the variation of PCB concentration in phytoplankton in 19 southern Swedish

321 lakes.[44] The present study performed in Lake Taihu implied a higher accordance of

322 PAHs with *Cyanophyta* and *Chlorophyta* and OCPs with *Bacillariophyta* and

323 *Cryptophyta* due to the species-specific uptake process, and then the internal uptake

324 mechanisms should be elucidated in the future study for evaluating the effects of

325 phytoplankton community on the fate of HOCs in aquatic ecosystem.

326 **Conclusion**

327 Both OCPs and PAHs are ubiquitous pollutants distributed in waters of Lake Taihu.

328 The residual levels of OCPs were similar to those reported in previous studies, while

329 PAHs were one to two orders of magnitude lower. The concentrations of OCPs and

330 PAHs in the summer water column were slightly higher than in winter due to such

331 physicochemical and biological parameters as the atmospheric deposition, uptake by

332 phytoplankton, discharge of wastewater, and resuspension of polluted sediments. The

333 primary study on the interactions between eutrophication and HOCs based on CCA

334 analysis demonstrated species-specific accumulation of hydrophobic contaminants by

335 phytoplankton, which would play a vital role in determining the biogeochemical

336 cycling of HOCs in eutrophic lakes. Furthermore, potential interactive processes

337 between HOCs and eutrophication should be included from a holistic approach for

338 water quality improvement in the future.

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416

417 **Figure captions**

418 Fig. 1 Sampling locations in Lake Taihu (fifteen samples distributed over the whole

419 lake from TH1 to TH15 included).

420 Fig. 2 Seasonal concentrations of OCPs (a) and PAHs (b) detected in surface waters

421 of Lake Taihu.

422 Fig. 3 RDA ordination biplots of PAH and OCP residues and predominant

423 phytoplankton communities in the Lake Taihu water column. (*Mic*: *Microcystis*; *Sce*:

424 *Scenedesmus*; *Ank*: *Ankistrodesmus*; *Sch*: *Schroederia*; *Aph*: *Aphanizomenon*; *Aul*:

425 *Aulacoseira*；*Ast*: *Asterionella*; *Cry*: *Cryptomonas*).

426

427   **Table 1** Winter and summer physicochemical and biological characters of water
428   samples from Lake Taihu. (Site designations refer to sampling station locations in Fig.
429   1)

| Sites	Winter					Summer				
	SS	DOC	*Cyanophyta*	Phytoplankton		SS	DOC	*Cyanophyta*	Phytoplankton	
	mg L^{-1}	mg L^{-1}	%	ind L^{-1}	mg L^{-1}	mg L^{-1}	mg L^{-1}	%	ind L^{-1}	mg L^{-1}
TH1	28.8	7.12	71.7	3.E+07	13.04	25.9	2.26	98.2	28.8	7.12
TH2	41.5	6.29	60.9	2.E+07	18.50	29.0	2.50	98.0	41.5	6.29
TH3	34.4	7.47	75.7	1.E+07	6.45	30.7	2.36	98.2	34.4	7.47
TH4	123.5	6.86	90.9	9.E+06	3.34	28.2	1.92	99.5	123.5	6.86
TH5	117.1	8.62	90.7	3.E+06	1.16	44.7	2.11	96.8	117.1	8.62
TH6	109.0	4.02	72.7	2.E+06	2.52	63.8	0.96	90.9	109.0	4.02
TH7	58.6	4.53	48.6	5.E+06	9.03	9.3	1.77	32.3	58.6	4.53
TH8	65.3	4.15	71.3	6.E+06	4.33	27.6	3.58	98.7	65.3	4.15
TH9	75.8	4.50	96.9	1.E+07	1.60	37.3	2.74	99.8	75.8	4.50
TH10	117.0	4.77	98.6	5.E+06	0.44	26.3	1.56	91.2	117.0	4.77
TH11	125.8	5.66	82.4	2.E+06	1.69	29.7	2.65	83.2	125.8	5.66
TH12	113.4	4.89	99.5	1.E+07	0.57	56.1	2.72	98.6	113.4	4.89
TH13	142.9	5.27	99.5	1.E+07	0.81	66.6	1.30	100.0	142.9	5.27
TH14	115.1	4.74	87.9	4.E+06	1.42	28.0	1.99	97.6	115.1	4.74
TH15	124.6	5.33	97.0	2.E+06	0.18	4.0	2.09	90.9	124.6	5.33

430
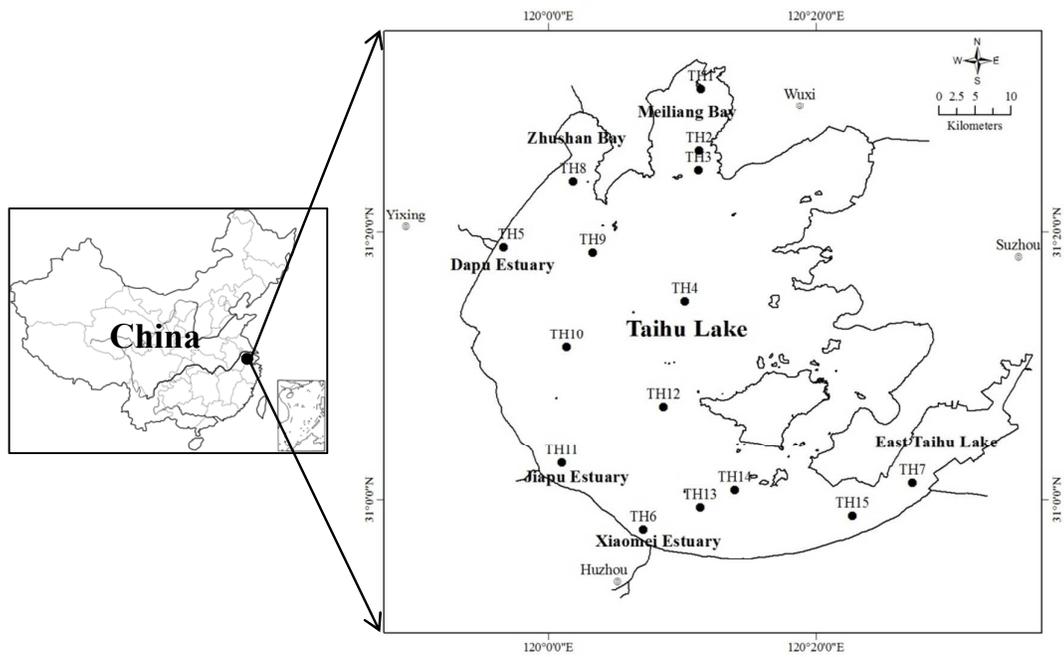
Fig. 1 Sampling locations in Lake Taihu (fifteen samples distributed over the whole lake from TH1 to TH15 included).
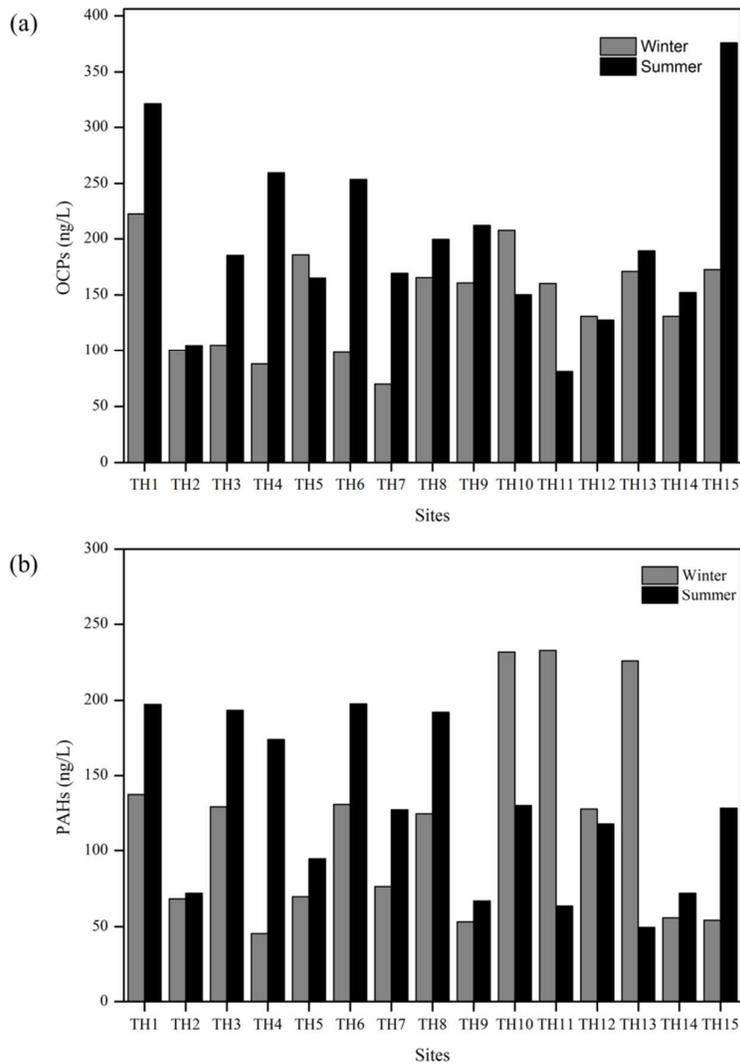
Fig. 2 Seasonal concentrations of OCPs (a) and PAHs (b) detected in surface waters of Lake Taihu.
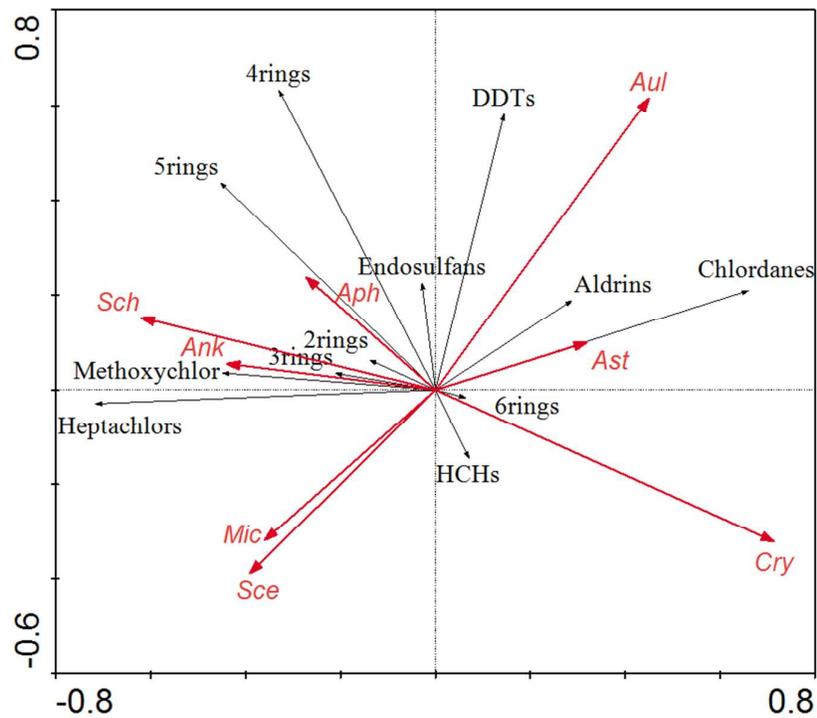
Fig. 3 RDA ordination biplots of PAH and OCP residues and predominant phytoplankton communities in the Lake Taihu water column. (*Mic*: *Microcystis*; *Sce*: *Scenedesmus*; *Ank*: *Ankistrodesmus*; *Sch*: *Schroederia*; *Aph*: *Aphanizomenon*; *Aul*: *Aulacoseira*；*Ast*: *Asterionella*; *Cry*: *Cryptomonas*).