Biogeosciences
Discussions

Open Access

EGU

1 # Are flood-driven turbidity currents hot-spots for priming
2 # effect in lakes?

3

4 **D. Bouffard[1] and M-E. Perga[2,*]**

5 [1]{Physics of Aquatic Systems Laboratory, Margaretha Kamprad Chair, EPFL-ENAC-IEE-
6 APHYS, CH-1015 Lausanne, Switzerland}

7 [2]{INRA-Université Savoie Mont Blanc, UMR 042 CARRTEL, Thonon les Bains, France}

8

9 Correspondence to: M-E. Perga (marie-elodie.perga@thonon.inra.fr)

10

11 **Abstract**

12 In deep stratified lakes, such as Lake Geneva, flood-driven turbidity currents are thought to
13 contribute to the replenishment of deep oxygen by significant transport of river waters
14 supersaturated with oxygen into the hypolimnion. The overarching aim of this study was to
15 test directly this long-standing hypothesis. It combines direct observational data collected
16 during an extreme flooding event that occurred in May 2015 with dark bioassays designed to
17 evaluate the consequences of riverborne inputs on the hypolimnetic respiration. The
18 exceptional precipitations of May 2015 caused floods with annual return time for the Rhône
19 River, the dominant tributary of Lake Geneva, and with 50-year return time for the Dranse
20 River, the second most important tributary. Sediment loaded river flows generated turbidity
21 currents plunging into the lake hypolimnion. The observed river intrusions contributed to the
22 redistribution of dissolved oxygen, with no net gain, when occurring in the lowermost
23 hypolimnetic layer. In the uppermost hypolimnion above the last deep mixing event the
24 intrusions coincided with a net oxygen deficit. Consistent with field observations, dark
25 bioassays showed that 1% to 50% substitution of riverine organic matter to deep (<200 m)
26 hypolimnetic water did not affect microbial respiration, while addition of 1 to 10% of riverine
27 water to the uppermost hypolimnetic waters resulted in a respiration overyielding, i.e. excess
28 respiration of both riverborne and lacustrine organic matter. The results of our study conflict
29 the hypothesis that flood-driven turbidity currents necessarily increase hypolimnetic oxygen

1 stocks in Lake Geneva. In contrast, they show that flood-driven turbidity currents can be
2 potential hot-spots for priming effect in lakes.

3 **1 Introduction**

4 In thermally stratified lakes, river water inflow occurs under two different modes. Under
5 normal (i.e dry-weather) flow conditions, the river water is injected at the interface between
6 the warm, upper layer (epilimnion) and the cold lower layer (hypolimnion), forming an
7 interflow in the upper thermocline (Fischer, 1979) or at the surface. Particularly during flood
8 events, high concentrations of suspended sediments in rivers increase the density of inflowing
9 waters and therefore generate turbid density currents following the slope and flowing into the
10 hypolimnion (i.e. hyperpycnal plume). Their dynamics are divided into three distinct stages
11 (Alavian et al., 1992; Cortés et al., 2014; Hogg et al., 2013). First, the river dense water
12 pushes the ambient lake water, until the resulting baroclinic pressure created by the local
13 density difference between the river and the lake water balance the force of the momentum
14 inflow. At this stage, the river flow plunges (plunging stage), then the flow continues to run
15 along the lake bed as an underflow (underflow regime). The flow eventually reaches the depth
16 of neutral buoyancy, separates from the lake bottom and intrudes into the lake (intrusion
17 stage).

18 Lake observations of such riverborne turbidity currents date back to the late XIXth century by
19 Forel in Lake Geneva (Forel, 1892), and more recent reports indicate that they occur in many
20 perialpine lakes such as Walensee (Lambert et al., 1976), Lake Geneva (Lambert and
21 Giovanoli, 1988), Lake Lucerne (Wüest et al., 1988), Lake Brienz (Finger et al., 2006), and
22 Lake Lugano (De Cesare et al., 2006). Marine underflows are also common features (Mulder
23 et al., 2003).

24 Besides their implications on the physical structure of subaquatic environments (Meiburg and
25 Kneller, 2010), underflows have been considered for their potential biogeochemical
26 consequences on lakes. For instance, in Lake Geneva, the long-standing hypothesis has been
27 that river intrusions could replenish deep oxygen, and this hypothesis was supported by
28 several, albeit indirect, field observations. Fahrni and Rapin (1986) compiled the densities of
29 Lake Geneva hypolimnetic waters and those of the Rhône River over seven years and
30 suggested that some of the time-periods prone to river underflows coincided with partial
31 oxygen replenishment in the deepest layers of the lake's central area (long-term monitoring
32 station SHL2). Meybeck et al. (1991) pointed out relatively frequent and important oxygen

1 and silicate anomalies occurring close to the sediment-water interface at the reference

2 monitoring station SHL2 and invoked riverborne underflows along with alternative

3 mechanisms (accumulation of turbid, cold water on lake banks after severe storms) as being

4 responsible for these anomalies.

5 Lake Geneva has been suffering from deep water hypoxia since its early eutrophication in the

6 late 1950s (Jenny et al., 2014). Based on the hypothesis of oxygen-rich river water intrusions

7 (Meybeck et al., 1991), underflows following episodes of heavy rainfalls are expected to

8 counteract hypoxia by supplying oxygen to the stratified lake hypolimnion (Jenny et al,

9 2014). Yet, numerous hydroelectric dams have been constructed on the course of the Rhône

10 River, leading to a shift in the seasonal discharge pattern (Loizeau and Dominik, 2000).

11 Consequently, large floods mostly occurring in summer were reduced in amplitude and

12 frequency, also leading to a decrease in sediment input by at least a factor 2 (Loizeau and

13 Dominik, 2000). This summer discharge and bulk plume density reduction impacted the

14 occurrence of underflows along the lake bottom and is thought to aggravate deep water

15 hypoxia. However, despite their crucial functional implications on hypolimnetic oxygen

16 concentrations, the consequences of turbidity currents, and of their further decrease in

17 occurrence, have actually never been investigated directly.

18 Condition for the riverborne currents to reach the bottom of the central basin is that the

19 hyperpycnal plume does not get diluted by lake water as it advances within a lake (Turner,

20 1986). Yet, sediment loaded underflows, as those expected during floods, differ from

21 underflows observed during regular river regime, whose density is only controlled by

22 temperature. While the latter typically intrudes in the metalimnion as recently revealed by

23 isotope-tracing (Halder et al., 2013), the former undergoes a dynamic density change,

24 generating multiple underflows and therefore increasing the overall dilution (Cortés et al.,

25 2014). In addition, underflows might primarily act on the hypolimnion by pushing deep

26 waters upwards (Wüest et al., 1988), therefore redistributing oxygen within deep layers, rather

27 than contributing to a net oxygen gain, as observed in nearby Lake Lugano (De Cesare et al.,

28 2006).

29 Besides these positive and null hypotheses, an alternative, metabolic hypothesis is formulated

30 here based on the observed impact of floods on the carbon budgets of some lakes around the

31 world. A common opinion is that the bulk of the organic carbon that enters surface water is

32 refractory and may poorly contribute to bacterial metabolism, in comparison to autochthonous

1   sources (Moran and Hodson, 1990). However, the composition and hence quality of dissolved

2   organic matter entering lakes depend on the watershed coverage, land-use, climate and

3   hydrology (Alvarez-Cobelas et al., 2010) and might also be highly variable during the year

4   (Berggren et al., 2009). Increased inputs of fresh organic material during stormwaters and

5   spring floods (Agren et al., 2008; Dhillon and Inamdar, 2013; Raymond and Saiers, 2010)

6   have been shown to push lake metabolic balances further into heterotrophy (Klug et al., 2012;

7   Sadro et al., 2011; Tsai et al., 2008), either by decreasing surface primary production through

8   lower water transparency or by stimulating bacterial respiration through addition of labile,

9   terrestrial organic matter (Johengen et al., 2008; Ojala et al., 2011; Vachon and Giorgio,

10   2014). Considering the size of Lake Geneva, it is unlikely that floods may affect the whole

11   ecosystem metabolism (Vachon and Giorgio, 2014). Yet, since riverborne intrusions are rich

12   in organic matter in Lake Geneva, they could locally be hotspots for bacterial respiration that

13   would counteract the net oxygen inputs in the hypolimnion. In this metabolic hypothesis,

14   riverborne intrusions could cause a null or even a negative effect on hypolimnetic oxygen

15   concentrations.

16   The overarching aim of this paper was, therefore, to study the net oxygen effect of flood

17   driven riverborne intrusions in Lake Geneva. The study combined direct observational data

18   collected during an extreme flooding event in May 2015 while the lake was already stratified

19   supported with an experimental test designed to evaluate the consequences of river water

20   inputs on the hypolimnetic respiration.

21   **2   Materials and methods**

22   **2.1   Field survey**

23   Lake Geneva is the largest lake of Western Europe in terms of volume (89 km^3), and depth

24   (309 m). 84 % of the water input originates from the two main rivers, the Rhône (75 %) and

25   the Dranse (9 %), both flowing into the western basin. The river discharge and water quality

26   are continuously monitored by the FOEN (Federal Office for the Environment, Switzerland

27   for the Rhône), the DREAL (Direction Régionale de l'Environnement, de l'Aménagement et

28   du   Logement,   for   discharge   of   the   Dranse   River

29   http://www.hydro.eaufrance.fr/stations/V0334010) and the Observatory of Alpine Lakes,

30   respectively (for the water quality of the Dranse, http://www6.inra.fr/soere-ola). Discharge

31   (hourly record) and water quality (two-weeks integrated sample) of the Rhône River are

4

Biogeosciences
Discussions

Open Access

EGU

1   monitored at Porte-du-Scex FOEN station 6 km upstream of the Rhône inflow. Discharge of

2   the Dranse is recorded at the Reyvroz hydrological station 20 km upstream of the Dranse inlet

3   to Lake Geneva while water quality surveys are performed close to the river delta.

4   While the Rhône river hydrological regime was originally of a typical glacier-type,

5   exploitation of the river flow for hydro-electrical production has substantially smoothed the

6   seasonal variability of water discharge over the latest 40 years, summer discharges being

7   currently only twice those observed in winter (Loizeau and Dominik, 2000). The average

8   annual discharge at Porte du Scex in May over 1935-2013 was 208 $m^3 s^{-1}$. The Dranse, which

9   is the second most important tributary of Lake Geneva, has a typical nival flow regime

10   (Meybeck et al., 1991), with maximum discharge in May (average discharge in May over

11   1906-2003: 39 $m^3 s^{-1}$).

12   The consequence of the heavy rain of May 2015 on physico-chemical lake properties was

13   investigated through a specific lake survey. 25 sites were sampled within one day with a

14   multi-parameter profiler (Sea&Sun Technology, CTD-90 multi-parameter probe), which

15   includes measurements of depth, temperature (T), conductivity, turbidity (Tu) and dissolved

16   oxygen concentration (O_2). The O_2 oxyguard (Clark type) is regularly calibrated with a long

17   term stability optode (Anderaa 4330F) but, in the present study, no drift in the O_2

18   measurements was expected during a single day survey.

19   The 25 sampling sites cover an area of ~200 km^2 over the Western basin. The sampling

20   design was specifically intended to sample the influence of the two main rivers and to

21   investigate the local influence of deep intrusions on O_2 concentrations. For this reason, all

22   sites were chosen with a local depth largely exceeding 100 m. Particular care was taken to

23   encompass the main active sub-lacustrine canyons of the Rhône and of the Dranse, as well as

24   their surroundings, and therefore to optimize the chance to probe intrusion plumes. The high

25   resolution CTD survey provided spatial information of O_2, Tu and T. Maps of hypolimnetic

26   properties were constructed with krigging interpolation method.

27   The net effect of river intrusion on the dissolved oxygen concentration was quantified by

28   comparing the O_2 profile within the intrusion layer to a theoretically-observed linearly

29   decreasing O_2 profile within this layer as typically observed in intrusion-free (undisturbed)

30   CTD profiles.

Biogeosciences
Discussions

Open Access

EGU

2.2 Oxygen consumption experiment

In order to test whether inputs of riverine organic carbon within the lake hypolimnion could drive significant oxygen consumption via microbial respiration, an incubation experiment was conducted in October 2015. On Oct 19th, 15 L of Lake Geneva hypolimnetic water were collected with a VanDorn bottle from 100 m and 200 m depths at SHL2, such as 20 L of water from the Dranse, at less than 1 km from its entrance into Lake Geneva. One L of each lake and river water was kept in a glass bottle for further analyses of Carbon (C), Nitrogen (N) and Phosphorus (P) contents. Concentrations in total and dissolved organic C (TOC, DOC) were measured with/without filtration on Whatman GF/F filters (0.7 μm nominal pore size), on a TIC/TOC analyser (OI Analytical). Nutrients were analysed by standard colorimetric methods (Association Française de Normalisation, 1990).

Pure lake and riverine waters, and mixed waters in which different percentages of lake water was substituted by riverine waters, were incubated in 280 mL hermetically-closed glass-bottles equipped with SP-PSt3 planar oxygen-sensitive spots (PreSens), according to the experimental design presented in Table 1. Triplicates of each sample were incubated in a temperature-controlled dark chamber at 10 °C (a realistic temperature for river and hypolimnetic conditions during the flood). Change in O_2 over time were measured using a PreSens Fibox 3 equipped with a fiber optic oxygen transmitter. Initial O_2 concentrations were measured 1-hour after the start of the incubation, once water temperature had stabilized at 10 °C. Thereafter, O_2, and corresponding oxygen consumption, was measured once or twice per day for four days. Results were analysed by ANCOVA using O_2 consumption ($O_{2,\ 0}$ - $O_{2,\ t}$) as the response variable, treatment as the factor and time as covariate, including interactions. Further comparisons between treatments or specific dates were performed using Student's T test.

3 Results

3.1 Field survey

For the 2015 spring flood events, heavy rainfalls over the Lake Geneva watershed started on May 1st until May 3rd (total rainfall > 100 mm over these three days). For comparison, the City of Bex (Switzerland), located 20 km upstream of Lake Geneva along the Rhône River, collected 101 mm of rain over these three days, a record that had last been observed in December 1916. The discharge of the Rhône increased from ~140 $m^3\,s^{-1}$ at the end of April up

1   to a maximum of 504 m^3 s^{-1} on May 4th (Fig. 1a). This discharge reached the 98th percentile of

2   the cumulative distribution of the Rhône discharges of 1976 - 2009, and corresponded to an

3   annual return time of the Rhône floods at the entrance of Lake Geneva (Fig. 1b). During the

4   flood, the Rhône water temperature was 8 °C and O$_2$ concentrations were 11 mgO$_2$ L^{-1} (data

5   source: FOEN, Switzerland). On the sampling date (May 7th), the Rhône discharges remained

6   elevated with a daily average of ~ 400 m^3 s^{-1}.

7   The heavy rainfalls of early May 2015 increased the flow of the Dranse, which was already at

8   its seasonal maximum, concurring in generating a flood of exceptional amplitude. Between

9   April 30st and May 4th, the discharge increased from 26 to 300 m^3 s^{-1}, right before the

10   monitoring station collapsed (Fig. 1a). This was a new record exceeding the previous

11   historical maximum discharge ever recorded at the Dranse hydrometric station, 229 m^3 s^{-1} on

12   September 22nd 1968. The estimated return time of this 2015 flood event for the Dranse was

13   50 years (Fig. 1c). The Dranse waters during the flood were highly turbid with concentration

14   of suspended matter reaching > 2000 mg L^{-1} (averaged concentration of 18 mg L^{-1} in 2014;

15   data from the observatory of large lakes, France). The suspended organic matter concentration

16   was 195 mg L^{-1} (annual average of 5 mg L^{-1} in 2014). DOC was twice the average annual

17   concentration (2.3 vs 1.2 mgC L^{-1}). Dissolved nutrient concentrations were moderate (PO$_4^{3-}$ =

18   10 µgP L^{-1}; NO$_3^-$ = 480 µgN L^{-1}). Although not regularly monitored, the highly torrential flow

19   of the Dranse remained close to saturated oxygen concentrations.

20   The high turbidity of the inflowing flood waters compared with the background turbidity

21   signal of Lake Geneva (< 5 FTU), was thereafter used as a tracer for intrusive waters within

22   the lake. CTD profiles for all surveyed stations are provided as Appendix data (Figure A1).

23   Of the 25 sites, more than 50 % had hypolimnetic turbidity peaks attributed to river intrusions

24   (Fig. 2). The use of turbidity as a proxy for riverborne waters was also validated by

25   temperature profiles showing a consistent increase of temperature in the turbid layers and

26   thereby suggesting that locally the density of the water is significantly affected by suspended

27   matters. This trend is clearly noticeable in BP18 located within the far Dranse underflow (Fig.

28   3a and b). The turbidity signal from the Rhône was restricted to the few stations located less

29   than 2 km downstream (BP8 and 9), while the turbidity current from the Dranse penetrated

30   much further within the lake, even reaching the reference monitoring station (SHL2 = BP18),

31   6 km downstream. The two northernmost stations BP2 and 3 were within the small but

32   noticeable underflow of the Veveyse River (Figure 2). Except for the stations closest to the

1   Rhône (BP8) and Dranse (BP21, 22 and 25), the underflow was split between two different

2   hypolimnetic layers: a very turbid underflow within the lower hypolimnion (below 110 m,

3   BP5, 7, 9 and 13 for the Rhône underflow, BP16 and 18 for the Dranse River), and less turbid

4   underflow between 50 and 110 m depths (BP2 and 29).

5   Undisturbed profiles typically indicated a similar trend in O_2 slope ($\Delta O_2 / \Delta depth$ = -0.22 mg

6   L^{-1} m^{-1}, SD 0.03 mg L^{-1} m^{-1}, based on 9 profiles, BP13, 4, 6, 11, 12, 14, 19 and 28, Figure A1)

7   in the lower hypolimnion defined as the region below the winter deep mixing maximum (110

8   m) and the layer of influence of the bottom (20 m above the sediment interface). The winter

9   deep mixing maximum at depth 110 m was also confirmed by a CTD profile carried out one

10   week earlier by the Observatory of Alpine Lakes, France (not shown). Although more

11   contrasted, the upper hypolimnion, i.e. between the thermocline depth and the winter deep

12   mixing depth (Fig. 2b), was characterized by a less steep O_2 slope (e.g. the upper hypolimnion

13   mixed three months earlier).

14   The net effect of the intrusion on the O_2 was first assessed by comparing intrusion-affected

15   and nearest intrusion-free CTD profiles (Fig. 3). Surprisingly, at no sampling site did the

16   turbidity peak match with a local maximum in O_2 that could compare to the lens anomalies

17   reported by Meybeck et al. (1991). Instead, the depths of the turbidity peak coincided with a

18   disruption of the background decreasing trend in O_2 as a function of depth, clearly noticeable

19   for instance in the comparison of the O_2 profiles at BP18 (affected by the Dranse) and BP16

20   (not affected, Fig. 3a and b). Such a reduction of O_2 vertical gradients as recorded at BP5, 8, 9

21   and 18, suggests the formation of a mixed layer due to the increased momentum within the

22   underflow. Depth-averaged O_2 in the interval 30 to 270 m at BP16 and BP18 were identical

23   (7.0 mgO_2 L^{-1}) and, thereby, supported the hypothesis that the studied intrusion and related

24   extreme flood event had no net effect on the O_2 concentration but rather efficiently mixed

25   turbid-affected hypolimnetic layers. Although the trend was not as clear as in the middle of

26   the lake, similar conclusions arose from the comparison of CTD profiles carried out near the

27   Rhône at BP8 and BP12 (Fig. 3b and c). More information on the Rhône intrusion is provided

28   as Appendix data (Figure A2).

29   At stations BP2, 3, 21, 22, 25, and 29, turbid layers above 110-m depths even coincided with

30   a decrease in O_2 concentration. The drop in O_2 at BP2 in the turbid layer between 58 m and 86

31   m is a stunning example with a decrease in O_2 of ~0.3 mgO_2 L^{-1}. O_2 concentration decline

1   within turbid layers was also observed near the Dranse at BP21, 22 and 25 (Fig. 4). Although

2   all three stations were affected by the Dranse underflow, turbid intrusion was observed at

3   different depths. While O_2 of the three stations was highly comparable at depths unaffected by

4   the turbid flow (15 - 50 m), their O_2 profiles diverged at depths affected by the turbidity

5   current. Below 50 m, O_2 concentration at BP21 dropped as the turbidity increased, while O_2

6   concentration at BP22 and BP25 remained higher and similar between 50 m and 70 m. Below

7   70 m, O_2 concentration at BP22 dropped as turbidity increased and last, the turbidity intrusion

8   at 90 m depth in BP 25 coincided with the collapse of the three O_2 profiles (e.g. O_2 drop at

9   BP25). Surprisingly, below 110 m, O_2 profiles remains similar at the three stations

10   independently of turbidity values.

11   The difference in depth-averaged O_2 between measured profile and associated linear fit

12   through the turbid layer provided a first order parameterization of the net O_2 effect of the

13   intrusion. Note that due to the spatial heterogeneity in such large system, it was impossible to

14   define a single reference profile valid for the entire lake. Furthermore, the change in O_2 slope

15   at the winter deep mixing maximum (110 m) precluded the use of this linear fitting method

16   for any intrusion encompassing this layer (i.e. BP 21, 22 and 25) although they clearly

17   showed evidence for oxygen depletion within the turbid layer (see above). We therefore

18   restricted this analysis to intrusions located in the upper part of the hypolimnion (between the

19   thermocline and the winter deep mixing maximum) or intrusions located below this winter

20   deep mixing maximum. Relative changes in O_2 in the turbid layers flowing within the lowest

21   hypolimnion (> 110 m depth) were not significant (-0.07 g m^{-3}, SD 0.05 g m^{-3}, t = -2.50, df =

22   4, p-value = 0.066 ; Fig. 6). Net oxygen effects associated turbid layers flowing within the

23   upper hypolimnion were more variable (-0.19 g m^{-3}, SD 0.16 g m^{-3}), but they were, on

24   average, significantly negative (t = -3.68, df = 7, p-value = 0.007), attesting of an actual

25   oxygen debt at these lower depths.

26   **3.2   Oxygen consumption experiment**

27   The experiment was designed *a posteriori* in order to explain observed differences in the

28   oxygen net effect of the Dranse intrusion between the upper and the lower hypolimnion

29   (above and below 110 m depth). In October 2015, DOC concentrations in the lake

30   hypolimnion and in the river were very similar (0.80 mgC L^{-1} at 100 m depths, 0.70 mgC L^{-1}

31   at 200 m depths and 0.75 mgC L^{-1} in the Dranse waters). Particulate organic carbon

32   concentrations were low (< 0.10 mgC L^{-1}). DOC in the Dranse waters during the experiment

1   were about three times lower than those observed during the flood but, more importantly,

2   DOC concentrations were highly comparable between dilution conditions. As a result,

3   differences in O_2 consumption between treatments cannot be driven by initial differences in

4   carbon contents.

5   Dissolved nutrient concentrations were very low in the Dranse at the time of collection.

6   Orthophosphate concentrations were half those recorded during the flood (5 µgP L^{-1}) while

7   nitrate concentrations were more similar (580 µgN L^{-1}). Orthophosphate concentrations at 100

8   m and 200 m depth were very comparable to those recorded during the flood (13 and 29 µgP

9   L^{-1} respectively at both dates) while nitrate concentrations were slightly lower (620 and 560

10 µgN L^{-1} in October, compared to 670 and 630 µgN L^{-1}, in May 2015).

11 Beyond 86 or 92 hours of incubations, some treatments (D100%, L200-100%, L200-99%)

12 presented a second phase of increased oxygen consumption that could indicate the start of

13 nitrification processes, i.e. oxygen consumption independent from aerobic mineralization. In

14 order to avoid any potential bias due to nitrification, final oxygen consumption values are

15 considered at 68 h of incubation. O_2 consumption over the first 68h was significantly different

16 between treatments (ANCOVA $F_{7,48}= 39$, $p < 2.10^{-16}$) and time ($F_{2,48} = 33$, $p = 8.10^{-10}$), with a

17 high consumption rate within the first 20 h, and a relative stabilization thereafter (Fig 6, a b).

18 O_2 consumption was the highest for the Dranse water, reaching 2.5 mgO_2 L^{-1} after 68 h, while

19 final values of O_2 consumption were significantly lower for the lake waters, and within Lake

20 Geneva waters, O_2 consumption was higher at 200 m than 100 m depth (0.9 mgO_2 L^{-1} and 0.5

21 mgO_2 L^{-1} respectively, t = 4.0, *p = 0.02*).

22 Dilution of Lake Geneva water at 200 m depth with water from the Dranse (L200-100%,

23 L200-99%, L200-90% and L200-50%) did not significantly affect the dynamics of O_2

24 consumption over time (ANCOVA, $F_{treatment\ 3,84} = 2.0$, p = 0.10, Fig 6a. a). O_2 consumption

25 between treatments were therefore not significantly different after 15 h or beyond of

26 incubation of 200 m deep water. In contrast, O_2 consumption in lake water collected at 100 m

27 depths was higher for a treatment with 1-10% of Dranse water added, as compared to the non-

28 diluted samples (ANCOVA, $F_{treatment\ 2,66} = 96$, $p < 2.10^{-16}$). From 15 h of incubation and

29 beyond, O_2 consumption in samples incubated with 1-10% of Dranse water was significantly

30 (25 - 150%) higher than for undiluted samples, although initial carbon content was similar

31 between all treatments (Fig. 6b).

1   **4   Discussion**

2   **4.1   River intrusions during the flooding event**

3   Considering the extreme intensity of the observed rain event and subsequent river discharges

4   we expected the flood-induced turbidity current to be heavy enough to trigger an underflow

5   along the lake bed and therefore reach the deepest layers of the water column. However, no

6   clear signatures of a bottom following underflow could be observed for this specific event.

7   We estimated the sediment load in the Rhône river during the flood event by fitting the

8   relationship between river discharge, Q, and sediment load, C with a power law $C = aQ^b$ as

9   suggested in Loizeau and Dominik (2000) and Mulder et al. (2003). Our best fit for the 50

10   years of measurements resulted in $a = 5.7 \times 10^{-4}$ and $b = 2.36$ (see Appendix data, A3) which

11   is in good agreement with previously estimated relationships (Loizeau and Dominik, 2000).

12   Based on this relationship, the resulting estimated sediment load at the flood paroxysmal

13   phase reached 1.4 kg m^{-3} (or g L^{-1}). Assuming that sediment load was predominantly made of

14   Quartz ($\rho_{sed} = 2700$ kg m^{-3}), the density of the Rhône river was estimated as $\rho_{R,tot} = \rho_w(S, T)$

15   $+(1-\rho_{sed} / \rho_w(S, T)) aQ^b = 1000.7$ kg m^{-3}, where $\rho_w(S, T)$ is the density of the water depending

16   on the temperature and salinity (Chen and Millero, 1986). This value was slightly lower than

17   the density of the lake water at the deepest location (1001.4 kg m^{-3}) and did not account for

18   the later entrainment of lake water into the intrusion. Similar estimates for the Dranse

19   provided $\rho_{R,tot} = 1001.2$ kg m^{-3} assuming the same river temperature than for the Rhône.

20   Similar first order calculation suggests that the lower part of the intrusion stopped at ~160 m

21   for Rhône water and at ~250 m for Dranse, water which is in very good agreement with the

22   observations. Our results therefore confirm that Rhône discharge with annual return time is

23   actually plunging. However, the underflow may find its equilibrium density in the

24   hypolimnion and further evolve as an intrusion, rather than a true hyperpycnal current, for

25   which much higher discharges might be required. Recent observations of a strong turbidite on

26   Lake Geneva (Corella et al., 2014) were, for instance, interpreted as the result of a major

27   underflow and resulting landslides in October 2000 with an extremely strong Rhône discharge

28   of nearly 1400 m3 s^{-1} (return time 300 years) and a sediment load of > 9 kg m^{-3}. Hence,

29   although Lambert and Giovanoli (1988) recorded 11 underflows associated with elevated or

30   rapidly changing discharge in the Rhône canyon, ~2.5 km away from the river mouth over a

31   short three months period in summer 1985, we could not time correlate any of these

32   underflows to any of the O$_2$ anomalies studied in the same year in the deepest 50 m of the

1 lake (269 - 309 m) by Meybeck et al (1991). This suggests that none of these underflows were

2 ultimately strong enough to travel far into the lake. Alternatively the previously postulated

3 relationship between O_2 anomalies and Rhône underflows as suggested by Meybeck et al.

4 (1991) is questionable as our observation shows that a one-year return time discharge rate

5 triggers a plunging underflow that will quickly degenerate into a deep intrusion. Underflows

6 evolving up to the centre of the lake require strong discharge with return times longer than

7 one year and are therefore infrequent. Due to its closer location to the lake centre, the Dranse

8 is more likely to affect the lake centre (BP18, SHL2) with interflow (present study) or

9 underflow (Meybeck et al. 1991).

10 **4.2 Consequences of river intrusions on hypolimnetic oxygen concentrations**

11 Overall, the dataset presented herein rejected the hypothesis of a net oxygen gain due the river

12 intrusions in Lake Geneva during this important flooding event.

13 When comparing our results to those that supported this original hypothesis, it seems that the

14 net effect of river intrusions on hypolimnetic oxygen concentrations of Lake Geneva varies

15 depending on the properties of the intrusion flow. In May 2015, neither the Dranse nor the

16 Rhône generated an underflow plunging to the lake bottom and we cannot exclude that such

17 hyperpycnal flows could indeed contribute to deep water oxygen replenishment. Instead, we

18 observed two types of intrusions, i.e. in the upper hypolimnion that had been previously

19 mixed during the preceding winter, and in the lower hypolimnion. Intrusions in the lower

20 hypolimnion acted essentially through their mixing momentum and partly redistributed

21 oxygen within the mixed layers with, yet, no net benefit. Intrusions above 110 m depths

22 consistently generated local oxygen depletion. Before the flood, O_2 concentrations in the

23 upper hypolimnion were higher than in the lower hypolimnion (9.5 mgO_2 L^{-1} and < 7 mgO_2 L^{-1}

24 respectively) because the winter mixing did not reach deeper than 110 m that year. The O_2

25 concentrations in the Rhône were as high as 11 mg L^{-1} and we assumed that the Dranse waters

26 were also slightly supersaturated. It is then unlikely that the observed oxygen depletion in the

27 turbidity current observed in the upper hypolimnion directly resulted from intrusions of O_2-

28 depleted river waters. Instead, they point to increased oxygen consumption in the uppermost

29 turbidity current, for which the metabolic consequences of the riverborne inputs would have

30 taken over its physical, mixing effect.

1   These observations suggested (i) that respiration of diluted, riverborne organic matter in the

2   hypolimnion had a significant effect on oxygen concentrations and (ii) that the contribution of

3   respiration varied between the upper and lowermost hypolimnion. The oxygen consumption

4   experiment that we designed, *a posteriori*, aimed at testing whether such assumptions were

5   reasonable. Although O_2 and DOC concentrations in the hypolimnetic waters were likely to

6   be relatively similar in October to those right before the flood, they were undoubtedly quite

7   different for the river waters. However, this experiment did not intend to mimic conditions

8   during the flood but instead to investigate the variability of the metabolic processes in the

9   different hypolimnetic layers.

10   **4.3   Hypolimnetic respiration of riverborne organic matter**

11   Microbial respiration for the Dranse water was initially three times those of the lake

12   hypolimnetic waters, for similar DOC concentrations. In order to best reproduce processes

13   occurring during the river intrusion in the lake, we did not filter water to remove plankton

14   before incubations, in contrast to experiments aiming at separating bacterial and planktonic

15   respiration rates (Warkentin et al., 2007). It is therefore likely that the respiration was higher

16   in the Dranse water samples because it included both bacterial and autotrophic planktonic

17   components compared to samples from the dark hypolimnetic layers in which the sole

18   microbial heterotrophs shall be present.

19   However, considering a respiratory quotient of 0.82 (Williams and del Giorgio, 2005),

20   consumed O_2 in the Dranse river samples after 68 h (i.e. 78 $\mu molO_2 \, L^{-1}$, i.e. 1.14 $\mu molO_2 \, L^{-1}$

21   h^{-1}) would correspond to the oxidation of 0.75 $mgC \, L^{-1}$, i.e. > 90 % of TOC. These values of

22   short –term oxygen consumption rates belong to the upper end of the range reported for lakes

23   and streams by Berggren et al. (2012). They attest of an important short-term labile pool of

24   DOC (*sensu* Guillemette and del Giorgio (2011)) in the river waters (low-molecular weight,

25   relatively young DOC, (Agren et al., 2008)) but also of low bacterial growth efficiency due to

26   nutrient limitation in the oligotrophic conditions of the Dranse rivers (Cimbleris and Kalff,

27   1998; Wiegner and Seitzinger, 2004). River water samples were collected for the experiment

28   purposes at times of moderate hydrological loads and DOC as well as phosphate

29   concentrations in the river during the flood were much higher suggesting fast leaching of the

30   watershed soils (Agren et al., 2008). It is likely that river DOC during the flood was even

31   more labile, since it was mobilized and transported by rapid flush and fast transport of soil

32   organic matter (Agren et al., 2008; Bergström and Jansson, 2000).

1   In contrast, respiration recorded in the lake hypolimnetic waters corresponded with lower

2   oxidation rates (23 % and 42 % of TOC at 100 m and 200 m depths, respectively) for similar

3   initial organic carbon contents. Bacterial growth (including respiration, production, and

4   growth efficiencies) depends both on nutrient limitation and organic matter quality (Farjalla et

5   al., 2009). At these depths, microbial metabolism is less likely to be nutrient limited but

6   bacterial abundances are nevertheless usually low (around 10^5 cells ml^{-1}, S. Jacquet, pers.

7   comm.) suggesting low values for bacterial production. Besides, hypolimnetic waters of Lake

8   Geneva have long-residence times (time of ~20 years (Meybeck, 1970) and although most of

9   the lakes' hypolimnetic DOC might primarily originate from autochthonous primary

10   production, DOC aging through microbial reworking contributes to increasing its aromaticity

11   (Berggren et al., 2009) resulting in low bacterial growth efficiencies even without nutrient

12   limitation (Berggren et al., 2009). In that case, lower respiration values for hypolimnetic

13   waters suggested that lake DOC was semi-labile as compared to the Dranse DOC that might

14   be fresher and more readily available.

15   Nevertheless, higher oxygen consumption rates measured for the lower hypolimnion as

16   compared to its upper layer are surprising at first sight as they point to a higher availability for

17   lake DOC of greater depths. Such depths–related differences in C availability for microbial

18   metabolism are also consistent with the substitution assays showing that riverborne, labile

19   DOC inputs stimulated microbial respiration only for the supposingly C-limited samples, i.e.

20   the 100m depth lake water (Eiler et al., 2003). If DOC had the same sources in both

21   hypolimnetic layers, the greater water retention time would instead contribute in decreasing

22   DOC bioavailability with depth. Yet, DOC concentrations increases between the lower limit

23   of the mixed hypolimnion (110 m) to the lake bottom (309 m), from 0.7 to 0.8 mgC L^{-1}, as a

24   likely consequence of DOC remobilization from the sediment and accumulation in the

25   overlaying water column (Gonsior et al., 2013). Recent studies highlighted DOC release from

26   the sediment is a substantial source of labile DOC to the water column (Downing et al.,

27   2008), which could increase to the short-term labile pool of DOC in the unmixed

28   hypolimnion. While additional investigation on deep DOC quality would be required, fluxes

29   of sediment DOC to the unmixed deeper hypolimnetic layer could sustain higher respiration

30   rates as compared to the most superficial one for which microbial metabolism is the most

31   limited by organic matter quality.

Biogeosciences
Discussions

Open Access

EGU

4.4 Excess respiration in the mixed samples and overyielding

More surprisingly though, the stimulation of microbial respiration for the 100-m depth treatment was disproportionate as compared to the quantity of added labile OC. The substitution of 1 % of lake DOC by riverborne, more labile DOC almost doubled the respiration rate within 68 hours. Substituting 1 % of the lake water from 100 m depth with Dranse water is predicted to generate an excess O_2 consumption of 0.018 mgO_2 L^{-1} based on the respiratory values of the D-100% samples. However, the experimentally observed values were +0.5 mgO_2 L^{-1}, showing a clear overyield (*sensu* Farjalla et al, 2009). Similarly, substitution of 10 % of lake DOC increased oxygen consumption by 1 mgO_2 L^{-1} while proportionality suggested instead the value of 0.18 mgO_2 L^{-1}.

Such effects were documented by Farjalla et al. (2009) who observed that a mixture of fresh and aged DOC acted synergistically on the bacterial respiration rate (Farjalla et al., 2009) resulting in disproportionately increased rates compared to single substrates. We did not filter the Dranse water to remove microbes prior to incubations, with the purpose of more closely replicating realistic conditions within turbidity currents, and it is likely that we added an inoculum of river microbes to the mixture experiments. A greater microbial diversity in the mixture samples could favour co-metabolism on carbon compound decomposition and therefore the observed overyielding, as suggested by Farjalla et al. (2009). Yet, more recent investigations revealed that the initial microbial community composition has less impact of DOC use than the nature of DOC itself (Attermeyer et al., 2014). The microbial riverine inoculum might then not account for the totality of the enhanced decomposition of DOC in the mixture, while metabolic synergies in the microbial use of the different DOC qualities could also be involved (Fonte et al., 2013).

4.5 Experimental and observational conclusions on the effect of river intrusion on the hypolimnetic oxygen concentrations of Lake Geneva.

The May 2015 flood episodes did not trigger a true underflow. Therefore, we cannot generally exclude that underflows, in which very high turbidity limits the mixing of the water masses of the river and the lake, can finally replenish deep water oxygen. However, rough estimations confirmed that such truly underflow processes are far more rare than previously thought. Exceptional events that indeed replenish oxygen at the bottom of the lake might occur at decennial, rather than annual time scales. Our observations pointed to null or negative effect

1   of river intrusions on the deep water oxygen content of Lake Geneva. Rather than increasing

2   deep water oxygen concentrations these intrusions cause physical mixing of the deep

3   hypolimnion, i.e. redistributing oxygen over depth, or have a metabolic effect. The final

4   consequences on the benthic biota are unclear since the effect on O_2 might be transient and

5   shall not persist for long within the lake hypolimnion. As a matter of fact, the change in the

6   deep hypolimnion O_2 profiles due to the homogenization by the intrusion was poorly visible

7   during the monitoring survey performed at SHL2 a few days after the end of the flooding

8   event (11 May, data not shown). While the underlying mechanisms explained why the relative

9   contribution of physical mixing and metabolism varied with the intrusion depth, our

10   observational survey and bioassays highlighted that these intrusions provide interfaces where

11   riverine and lake organic matter are mixed and can act as biogeochemical hotspots. Since the

12   quantity of substituted DOC does not account for the excess oxygen consumption observed in

13   the L100-99% and L100-90% treatments, it means that more of the 100 m depth,

14   hypolimnetic DOC had been respired as a small fraction of more labile, riverborne, DOM was

15   substituted. Even if the role played by a potential inoculum of riverine microbes cannot be

16   ruled out, river intrusions in the upper hypolimnion resulted in an increase of both

17   autochthonous and allochthonous organic matter respiration. This mechanism by which a

18   small addition of labile organic matter stimulates the mineralization of less available organic

19   matter is referred as to 'priming effect'. Priming effect has been thoroughly investigated in

20   soils but evidence are still sought in aquatic systems (Bianchi, 2011; Catalán et al., 2015).

21   River plumes had been identified as sites prone to host priming effect, since they shall bring

22   in contact different sources of organic matters with varying quality (Bianchi, 2011; Guenet et

23   al., 2010). Even though several papers, along with the present results, have revised the long-

24   standing hypothesis of the recalcitrance of terrestrial organic matter (Guillemette and del

25   Giorgio, 2011; Roehm et al., 2009), investigations of aquatic priming effect are still based on

26   such hypothesis that lacustrine OM shall be the primer and terrestrial OM the "primed"

27   (Catalán et al., 2015).   In the present study, the vertical consideration of the intrusion

28   challenged our preconceived thoughts on the quality of aquatic DOC, showing that lacustrine

29   DOC was semi-labile, but of heterogeneous quality with depths, while river OM was

30   potentially acting as the primer. Overall, deep river intrusion in Lake Geneva might therefore

31   be potential hotspots for aquatic priming, fostering the mineralization of deep, less labile

32   lacustrine organic matter.

33

Biogeosciences

Open Access

EGU

Discussions

1

2   **Appendix**

3

4   A1. CTD profiles

5

6   A2. CTD transect from the Rhône mouth to the lake centre. These series of CTD profiles

7   suggest that the net O_2 effect of the Rhône intrusion is limited or null but instead efficiently

8   homogenized the O_2 in the hypolimnetic water affected by the intrusion (reduction of the O_2

9   gradient close to the Rhône River). Intrusion are associated with elevated Tu signal.

10

11   A3. Sediment rating curve of the Rhône River at La Porte-du-Scex from 1964 to 2015. Best

12   fit Concentration [mg L^{-1}] = aQ^b, with Q = discharge [m^3 s^{-1}], yields a = 5.7 x 10^{-4} (3.1 x 10^{-4},

13   8.3 x 10^{-4}) and b = 2.365 (2.29, 2.441) with the 95% confidence interval in parenthesis.

14

15   A4. GPS coordinate (geodetic datum CH1903+) of the CTD profiles carried out on May 7th

16   2015

17

18   **Author contribution**

19   Both authors contributed equally to the field work, the data analysis and the redaction of the

20   manuscript.

21

22

26

Biogeosciences
Discussions
Open Access
EGU

1   **References**

2   Agren, A., Berggren, M., Laudon, H. and Jansson, M.: Terrestrial export of highly

3   bioavailable carbon from small boreal catchments in spring floods, Freshw. Biol., 53(5), 964–

4   972, 2008.

5   Alavian, V., Jirka, G. H., Denton, R. A., Johnson, M. C. and Stefan, H. G.: Density currents

6   entering lakes and reservoirs, J. Hydraul. Eng., 118(11), 1464–1489, 1992.

7   Alvarez-Cobelas, M., Angeler, D. G., Sánchez-Carrillo, S. and Almendros, G.: A worldwide

8   view of organic carbon export from catchments, Biogeochemistry, 107(1-3), 275–293,

9   doi:10.1007/s10533-010-9553-z, 2010.

10   Attermeyer, K., Hornick, T., Kayler, Z. E., Bahr, A., Zwirnmann, E., Grossart, H.-P. and

11   Premke, K.: Enhanced bacterial decomposition with increasing addition of autochthonous to

12   allochthonous carbon without any effect on bacterial community composition,

13   Biogeosciences, 11(6), 1479–1489, 2014.

14   Berggren, M., Laudon, H. and Jansson, M.: Aging of allochthonous organic carbon regulates

15   bacterial production in unproductive boreal lakes, Limnol. Oceanogr., 54(4), 1333–1342,

16   doi:10.4319/lo.2009.54.4.1333, 2009.

17   Berggren, M., Lapierre, J.-F. and del Giorgio, P. A.: Magnitude and regulation of

18   bacterioplankton respiratory quotient across freshwater environmental gradients, ISME J.,

19   6(5), 984–993, 2012.

20   Bergström, A.-K. and Jansson, M.: Bacterioplankton production in humic Lake Örträsket in

21   relation to input of bacterial cells and input of allochthonous organic carbon, Microb. Ecol.,

22   39(2), 101–115, 2000.

23   Bianchi, T. S.: The role of terrestrially derived organic carbon in the coastal ocean: A

24   changing paradigm and the priming effect, Proc. Natl. Acad. Sci., 108(49), 19473–19481,

25   2011.

26   Catalán, N., Kellerman, A. M., Peter, H., Carmona, F. and Tranvik, L. J.: Absence of a

27   priming effect on dissolved organic carbon degradation in lake water, Limnol. Oceanogr.,

28   60(1), 159–168, 2015.

29   Chen, C.-T. A. and Millero, F. J.: thermodynamic properties for natural waters covering only

30   the limnological range1, Limnol. Oceanogr., 31(3), 657–662, 1986.

Biogeosciences
Discussions

EGU

Open Access

1   Cimbleris, A. C. and Kalff, J.: Planktonic bacterial respiration as a function of C: N: P ratios
2   across temperate lakes, Hydrobiologia, 384(1-3), 89–100, 1998.

3   Corella, J. P., Arantegui, A., Loizeau, J.-L., DelSontro, T., Le Dantec, N., Stark, N.,
4   Anselmetti, F. S. and Girardclos, S.: Sediment dynamics in the subaquatic channel of the
5   Rhone delta (Lake Geneva, France/Switzerland), Aquat. Sci., 76(1), 73–87, 2014.

6   Cortés, A., Fleenor, W. E., Wells, M. G., de Vicente, I. and Rueda, F. J.: Pathways of river
7   water to the surface layers of stratified reservoirs, Limnol. Oceanogr., 59(1), 233–250,
8   doi:10.4319/lo.2014.59.1.0233, 2014.

9   De Cesare, G., Boillat, J.-L. and Schleiss, A. J.: Circulation in stratified lakes due to flood-
10   induced turbidity currents, J. Environ. Eng., 132(11), 1508–1517, 2006.

11   Dhillon, G. S. and Inamdar, S.: Extreme storms and changes in particulate and dissolved
12   organic carbon in runoff: Entering uncharted waters?, Geophys. Res. Lett., 40(7), 1322–1327,
13   doi:10.1002/grl.50306, 2013.

14   Downing, B. D., Bergamaschi, B. A., Evans, D. G. and Boss, E.: Assessing contribution of
15   DOC from sediments to a drinking-water reservoir using optical profiling, Lake Reserv.
16   Manag., 24(4), 381–391, 2008.

17   Eiler, A., Langenheder, S., Bertilsson, S. and Tranvik, L. J.: Heterotrophic bacterial growth
18   efficiency and community structure at different natural organic carbon concentrations, Appl.
19   Environ. Microbiol., 69(7), 3701–3709, 2003.

20   Fahrni, H. P. and Rapin, F.: Modélisation du cycle du phosphore dans le Léman, Rapports sur
21   les études et recherches entreprises dans le bassin lémanique. Campagne 1985., 1986.

22   Farjalla, V. F., Marinho, C. C., Faria, B. M., Amado, A. M., Esteves, F. de A., Bozelli, R. L.
23   and Giroldo, D.: Synergy of fresh and accumulated organic matter to bacterial growth,
24   Microb. Ecol., 57(4), 657–666, 2009.

25   Finger, D., Schmid, M. and Wüest, A.: Effects of upstream hydropower operation on riverine
26   particle transport and turbidity in downstream lakes, Water Resour. Res., 42(8), 2006.

27   Fischer, H. B.: Mixing in Inland and Coastal Waters, Academic Press., 1979.

28   Fonte, E. S., Amado, A. M., Meirelles-Pereira, F., Esteves, F. A., Rosado, A. S. and Farjalla,
29   V. F.: The combination of different carbon sources enhances bacterial growth efficiency in
30   aquatic ecosystems, Microb. Ecol., 66(4), 871–878, 2013.

1 Forel, F. A.: Le Léman: monographie limnologique, F. Rouge., 1892.

2 Gonsior, M., Schmitt-Kopplin, P. and Bastviken, D.: Depth-dependent molecular composition
3 and photo-reactivity of dissolved organic matter in a boreal lake under winter and summer
4 conditions, Biogeosciences, 10(11), 6945–6956, 2013.

5 Guenet, B., Danger, M., Abbadie, L. and Lacroix, G.: Priming effect: bridging the gap
6 between terrestrial and aquatic ecology, Ecology, 91(10), 2850–2861, 2010.

7 Guillemette, F. and del Giorgio, P. A.: Reconstructing the various facets of dissolved organic
8 carbon bioavailability in freshwater ecosystems, Limnol. Oceanogr., 56(2), 734–748, 2011.

9 Halder, J., Decrouy, L. and Vennemann, T. W.: Mixing of Rhône River water in Lake Geneva
10 (Switzerland–France) inferred from stable hydrogen and oxygen isotope profiles, J. Hydrol.,
11 477, 152–164, doi:10.1016/j.jhydrol.2012.11.026, 2013.

12 Hogg, C. A. R., Marti, C. L., Huppert, H. E. and Imberger, J.: Mixing of an interflow into the
13 ambient water of Lake Iseo, Limnol. Oceanogr., 58(2), 579–592,
14 doi:10.4319/lo.2013.58.2.0579, 2013.

15 Jenny, J.-P., Arnaud, F., Alric, B., Dorioz, J.-M., Sabatier, P., Meybeck, M. and Perga, M.-E.:
16 Inherited hypoxia: A new challenge for reoligotrophicated lakes under global warming, Glob.
17 Biogeochem. Cycles, 28(12), 2014GB004932, doi:10.1002/2014GB004932, 2014.

18 Johengen, T. H., Biddanda, B. A. and Cotner, J. B.: Stimulation of Lake Michigan plankton
19 metabolism by sediment resuspension and river runoff, J. Gt. Lakes Res., 34(2), 213–227,
20 2008.

21 Klug, J. L., Richardson, D. C., Ewing, H. A., Hargreaves, B. R., Samal, N. R., Vachon, D.,
22 Pierson, D. C., Lindsey, A. M., O'Donnell, D. M., Effler, S. W. and others: Ecosystem effects
23 of a tropical cyclone on a network of lakes in northeastern North America, Environ. Sci.
24 Technol., 46(21), 11693–11701, 2012.

25 Lambert, A. and Giovanoli, F.: Records of riverborne turbidity currents and indications of
26 slope failures in the Rhone delta of Lake Geneva, Limnol. Oceanogr., 33(3), 458–468,
27 doi:10.4319/lo.1988.33.3.0458, 1988.

28 Lambert, A. M., Kelts, K. R. and Marshall, N. F.: Measurements of density underflows from
29 Walensee, Switzerland, Sedimentology, 23(1), 87–105, 1976.

Biogeosciences
Discussions

Open Access

1   Loizeau, J.-L. and Dominik, J.: Evolution of the Upper Rhone River discharge and suspended

2   sediment load during the last 80 years and some implications for Lake Geneva, Aquat. Sci.,

3   62(1), 54–67, doi:10.1007/s000270050075, 2000.

4   Meiburg, E. and Kneller, B.: Turbidity currents and their deposits, Annu. Rev. Fluid Mech.,

5   42, 135–156, 2010.

6   Meybeck, M.: Bilan hydrochimique et géochimique du Lac Léman, PhD Thesis, 1970.

7   Meybeck, M., Blanc, P., Moulherac, A. E. and Corvi, C.: Chemical evidence of water

8   movements in the deepest part of Lake Leman (Lake Geneva), Aquat. Sci., 53(4), 273–289,

9   1991.

10 Moran, M. A. and Hodson, R. E.: Bacterial production on humic and nonhumic components

11 of dissolved organic carbon, Limnol. Oceanogr., 35(8), 1744–1756, 1990.

12 Mulder, T., Syvitski, J. P., Migeon, S., Faugeres, J.-C. and Savoye, B.: Marine hyperpycnal

13 flows: initiation, behavior and related deposits. A review, Mar. Pet. Geol., 20(6), 861–882,

14 2003.

15 Ojala, A., Bellido, J. L., Tulonen, T., Kankaala, P. and Huotari, J.: Carbon gas fluxes from a

16 brown-water and a clear-water lake in the boreal zone during a summer with extreme rain

17 events, Limnol. Oceanogr., 56(1), 61–76, 2011.

18 Raymond, P. A. and Saiers, J. E.: Event controlled DOC export from forested watersheds,

19 Biogeochemistry, 100(1-3), 197–209, doi:10.1007/s10533-010-9416-7, 2010.

20 Roehm, C. L., Giesler, R. and Karlsson, J.: Bioavailability of terrestrial organic carbon to lake

21 bacteria: The case of a degrading subarctic permafrost mire complex, J. Geophys. Res.

22 Biogeosciences 2005–2012, 114(G3), 2009.

23 Sadro, S., Melack, J. M. and MacIntyre, S.: Depth-integrated estimates of ecosystem

24 metabolism in a high-elevation lake (Emerald Lake, Sierra Nevada, California), Limnol.

25 Oceanogr., 56(5), 1764–1780, 2011.

26 Tsai, J.-W., Kratz, T. K., Hanson, P. C., Wu, J.-T., Chang, W. Y., Arzberger, P. W., Lin, B.-

27 S., Lin, F.-P., Chou, H.-M. and Chiu, C.-Y.: Seasonal dynamics, typhoons and the regulation

28 of lake metabolism in a subtropical humic lake, Freshw. Biol., 53(10), 1929–1941, 2008.

29 Turner, J. S.: Turbulent entrainment: the development of the entrainment assumption, and its

30 application to geophysical flows, J. Fluid Mech., 173, 431–471, 1986.

1   Vachon, D. and Giorgio, P. A. del: Whole-Lake CO2 Dynamics in Response to Storm Events

2   in Two Morphologically Different Lakes, Ecosystems, 17(8), 1338–1353,

3   doi:10.1007/s10021-014-9799-8, 2014.

4   Warkentin, M., Freese, H. M., Karsten, U. and Schumann, R.: New and fast method to

5   quantify respiration rates of bacterial and plankton communities in freshwater ecosystems by

6   using optical oxygen sensor spots, Appl. Environ. Microbiol., 73(21), 6722–6729, 2007.

7   Wiegner, T. N. and Seitzinger, S. P.: Seasonal bioavailability of dissolved organic carbon and

8   nitrogen from pristine and polluted freshwater wetlands, Limnol. Oceanogr., 49(5), 1703–

9   1712, 2004.

10 Williams, P. J. and del Giorgio, P. A.: Respiration in aquatic ecosystems: history and

11 background, Respir. Aquat. Ecosyst. Oxf. Univ. Press Oxf., 1–17, 2005.

12 Wüest, A., Imboden, D. M. and Schurter, M.: Origin and size of hypolimnic mixing in

13 Urnersee, the southern basin of Vierwaldstättersee (Lake Lucerne), Schweiz. Z. Für Hydrol.,

14 50(1), 40–70, 1988.

15

16

Biogeosciences
Open Access
Discussions
EGU

1   **Figures captions**

2

3   Fig. 1. Hydrological characteristics of the flooding event of May 2015. a. Daily precipitation

4   and discharges of the two dominant tributaries of Lake Geneva, the Rhône and the Dranse

5   rivers over the months preceding and following the flood. The break in the discharge record

6   of the Dranse corresponds to the date at which the station collapsed. Cumulative distribution

7   of the Rhône discharges in 1976-2009 (b) and Dranse discharges 1957-2014 (c). Red lines

8   indicates the maximum discharges of the May 2015 event.

9

10   Fig. 2a. Spatial distribution of hypolimnetic turbidity (40 - 300 m depth) as a tracer for

11   flooding river intrusions. Identified stations are those for which profiles were provided in

12   Figure 3b. The inserted figure shows typical temperature and O_2 profiles for an undisturbed

13   station. Note the linear decrease of O_2 with depth in the lower hypolimnion (below the deep

14   winter maximum). GPS location of the CTD profiles is indicated in the Appendix information

15   (A4).

16

17   Fig. 3. Comparison of temperature, turbidity and O_2 depth-profiles for nearby stations, one

18   being undisturbed (dotted lines), and the other highly disturbed (continuous lines) by the

19   turbidity current of the Dranse (a,b); the Veveyse (c,d) and the Rhône rivers (d,e,).

20

21   Fig. 4. Comparison of turbidity (a) and O_2 (b) depth-profiles for the three stations close to the

22   Dranse river mouth. The shaded area corresponds to the upper hypolimnion, i.e. water layers

23   that have been mixed during the preceding winter.

24

25   Fig. 5. Net effect of the turbidity layer on O_2 concentrations calculated for intrusion above

26   and below the deep winter winter maximum (110 m).

27

28   Fig. 6. O_2 consumption in the bioassays. a. Bioassays conducted for the lake water collected in

29   the lowermost hypolimnion at 200m depth (100% 200 m), and with 1, 10% and 50%

1 substitution, respectively, with Dranse water (99% 200 m, 90% 200 m and 50% 200 m,

2 respectively, and compared for Dranse water only (D 100%). b. Bioassays conducted for the

3 lake water collected in the uppermost hypolimnion at 100 m depth (100% 100 m), with 1 and

4 10% substitution, respectively, with Dranse water (99% 100 m, 90% 100 m respectively), and

5 compared for the Dranse water (D 100%).

6

7

8

9

1

Table 1. Design of the incubation experiment.

3

Sample designation	Percent composition Lake Water		
	200m-depth lake water	100m-depth lake water	Dranse water
L200-100%	100%		
L200-99%	99%		1%
L200-90%	90%		10%
L200-50%	50%		50%
L100-100%		100%	
L100-99%		99%	1%
L100-90%		90%	10%
D-100%			100%

4

5

1

2

3   **Figures**

4

5

6   Figure 1

7

8
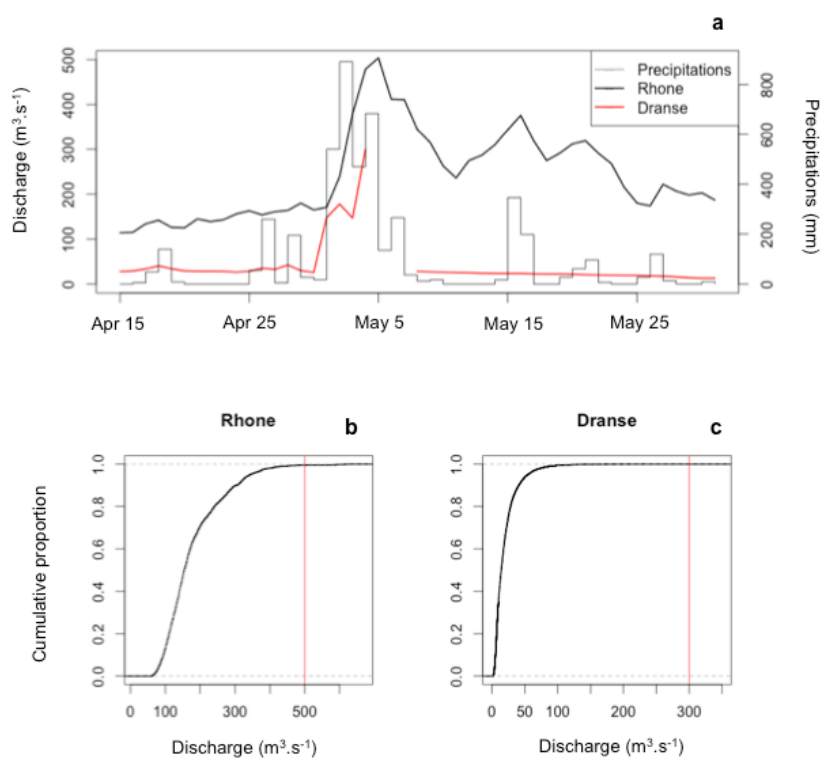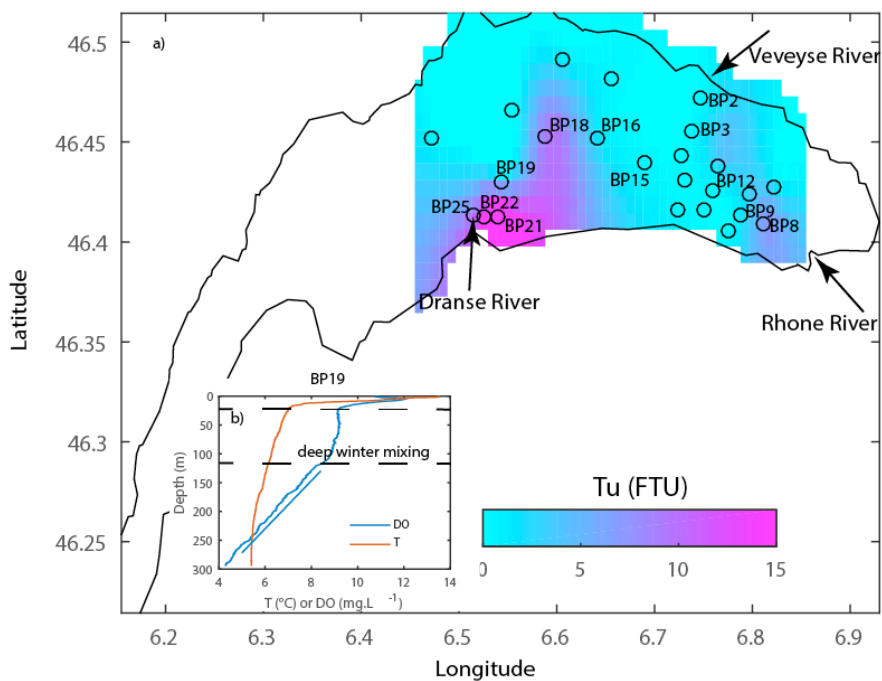
1

2   Figure 2

3

4

5

6

1

2
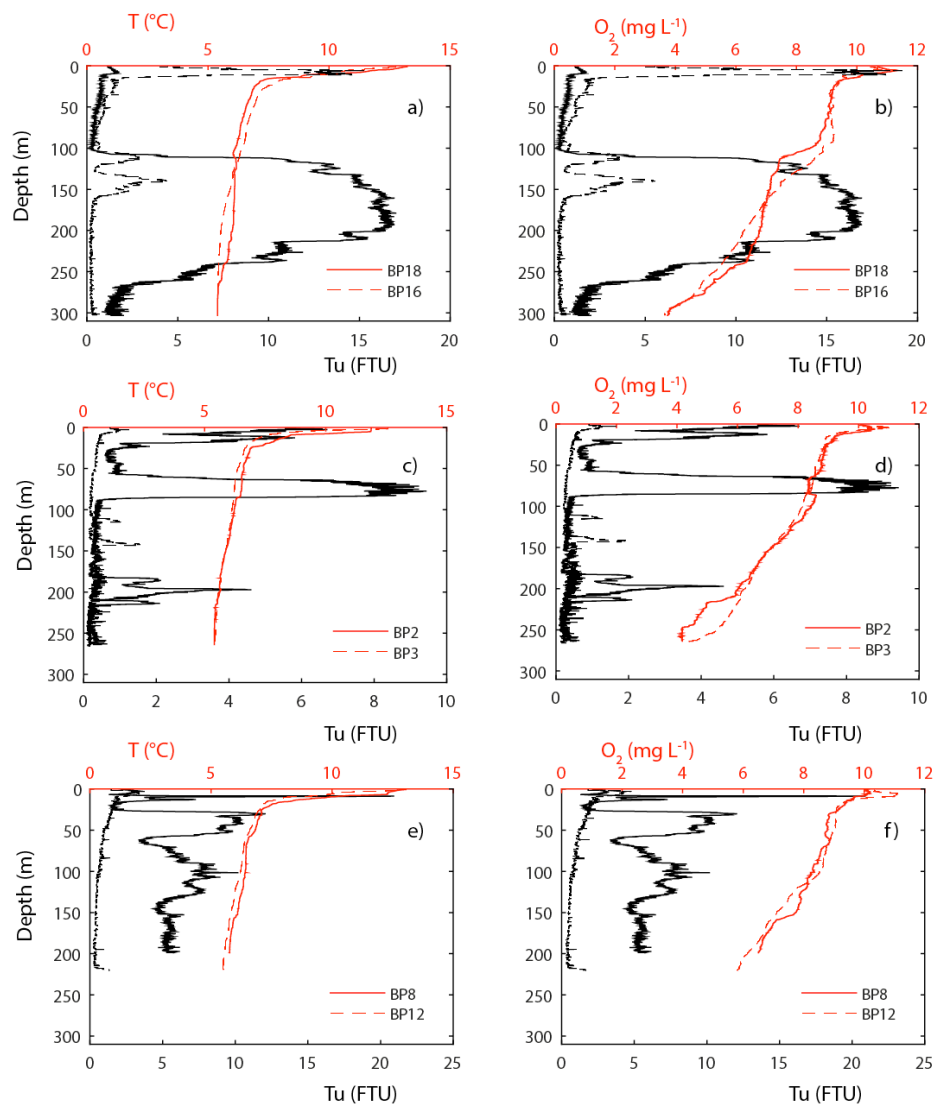
3

4   Figure 3

5

6

7

1

2
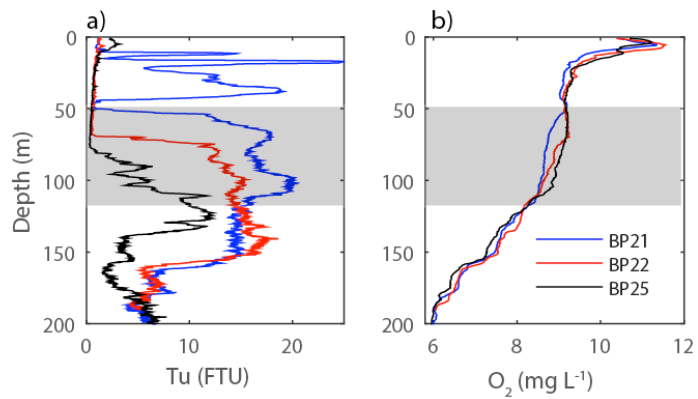
3

4   Figure 4

5

6

7

8

1

2

3

4

5   Figure 5
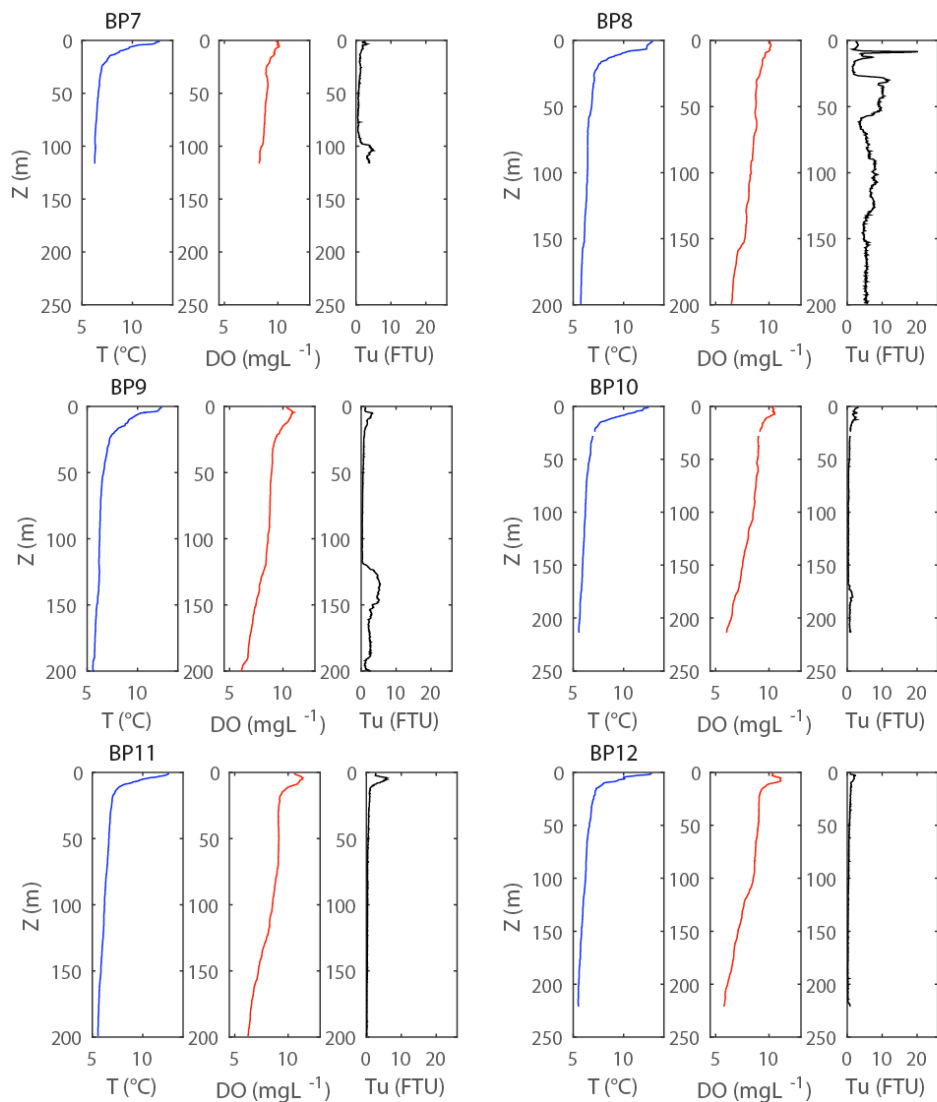
6

7
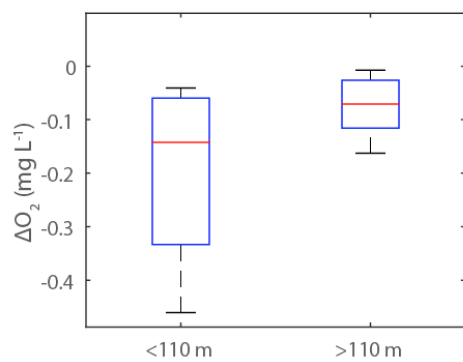
1
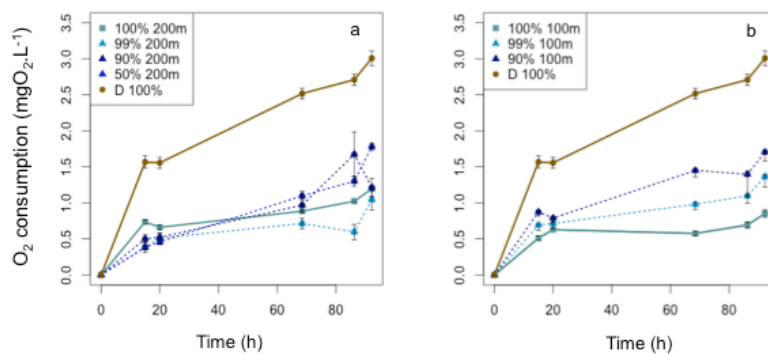
2

3   Figure 6
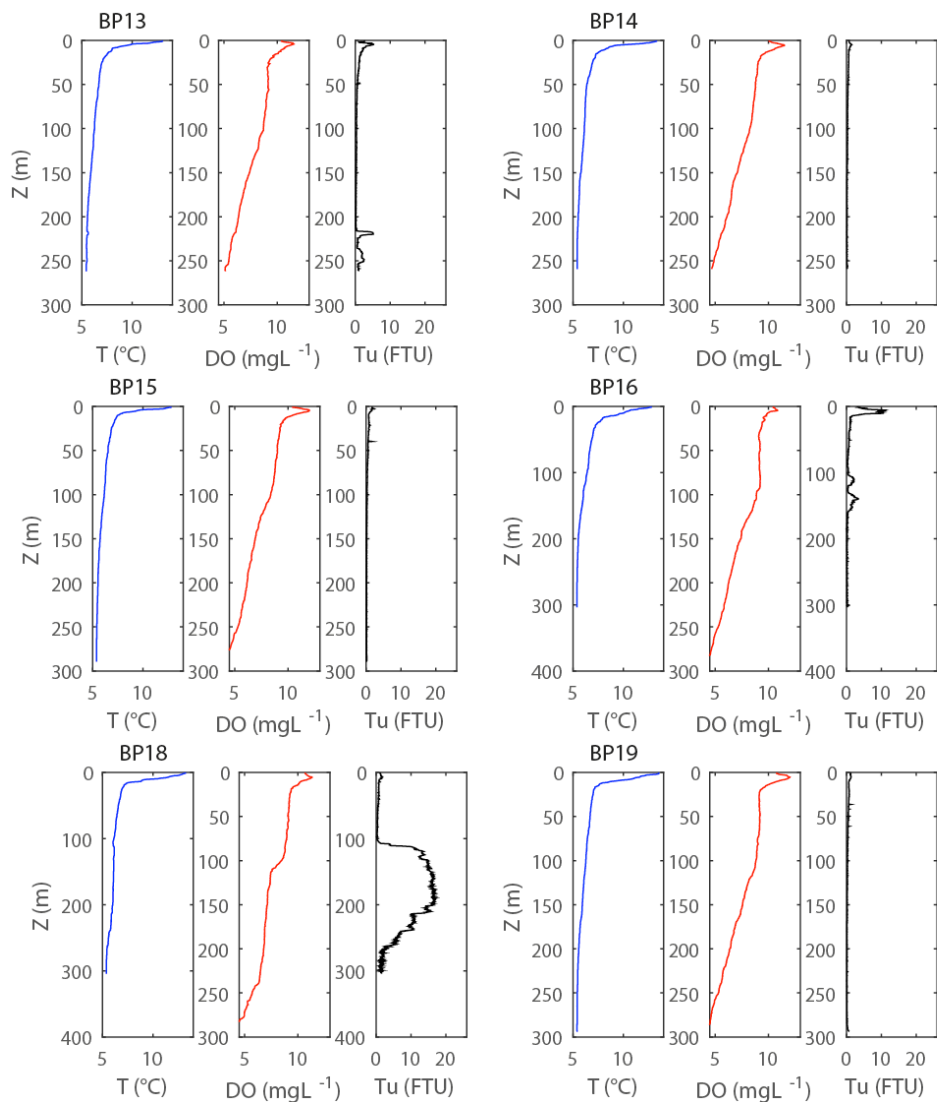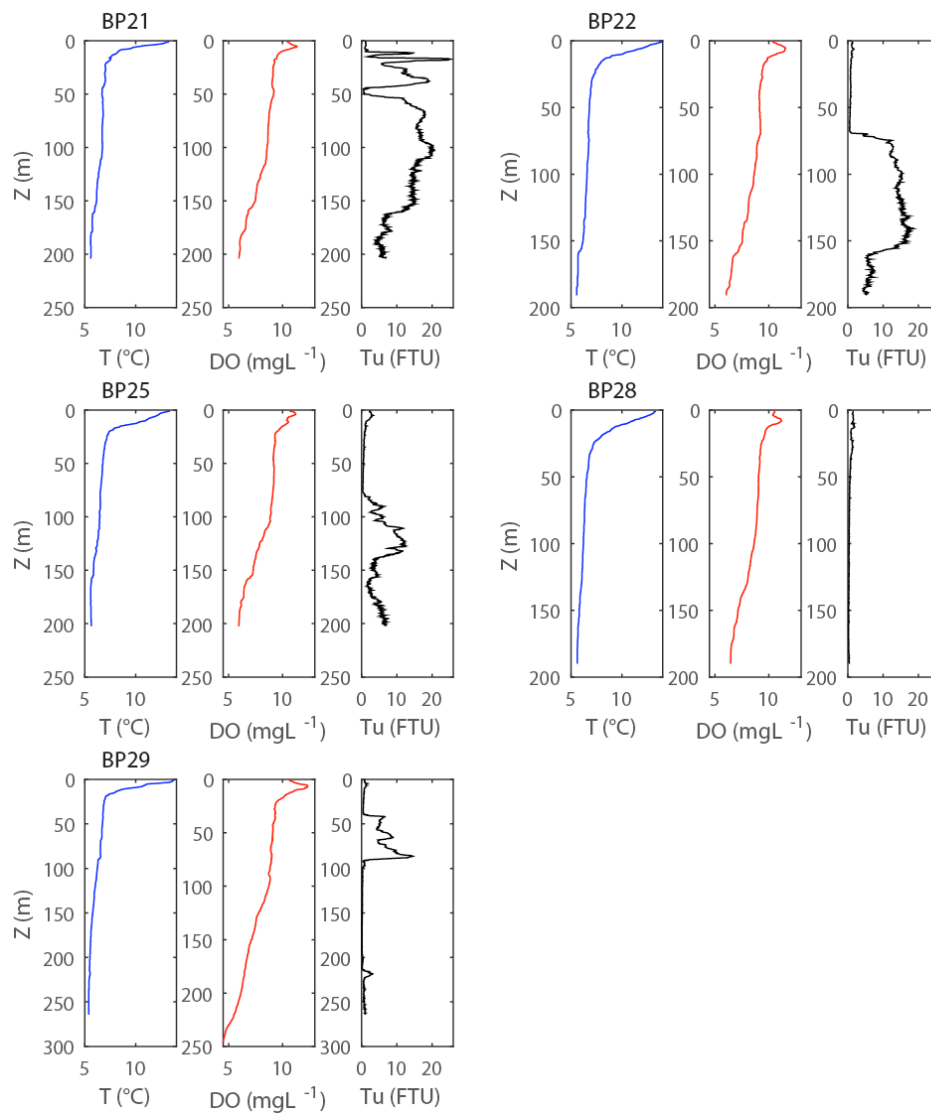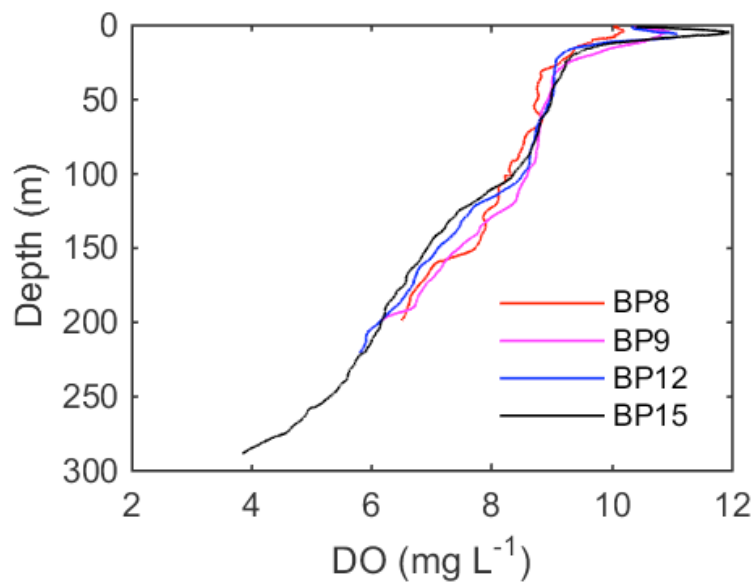
4

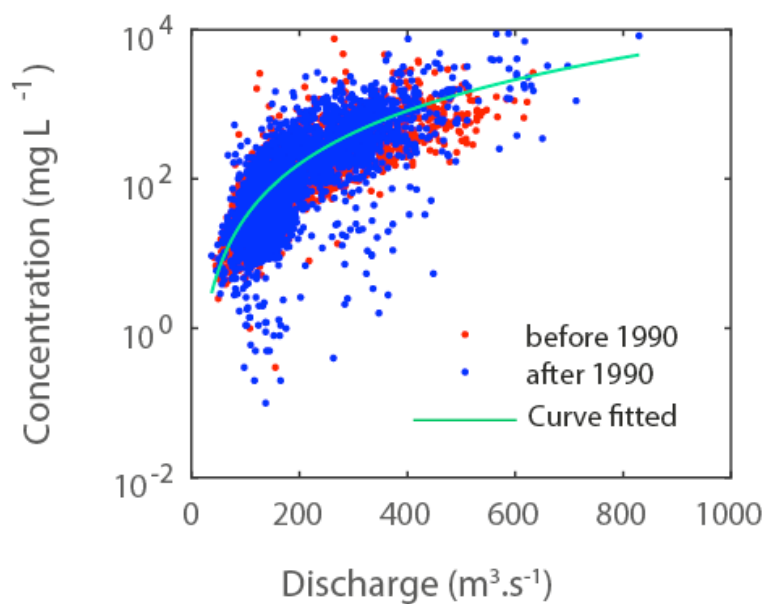1 **Appendix**

2 A1.

3
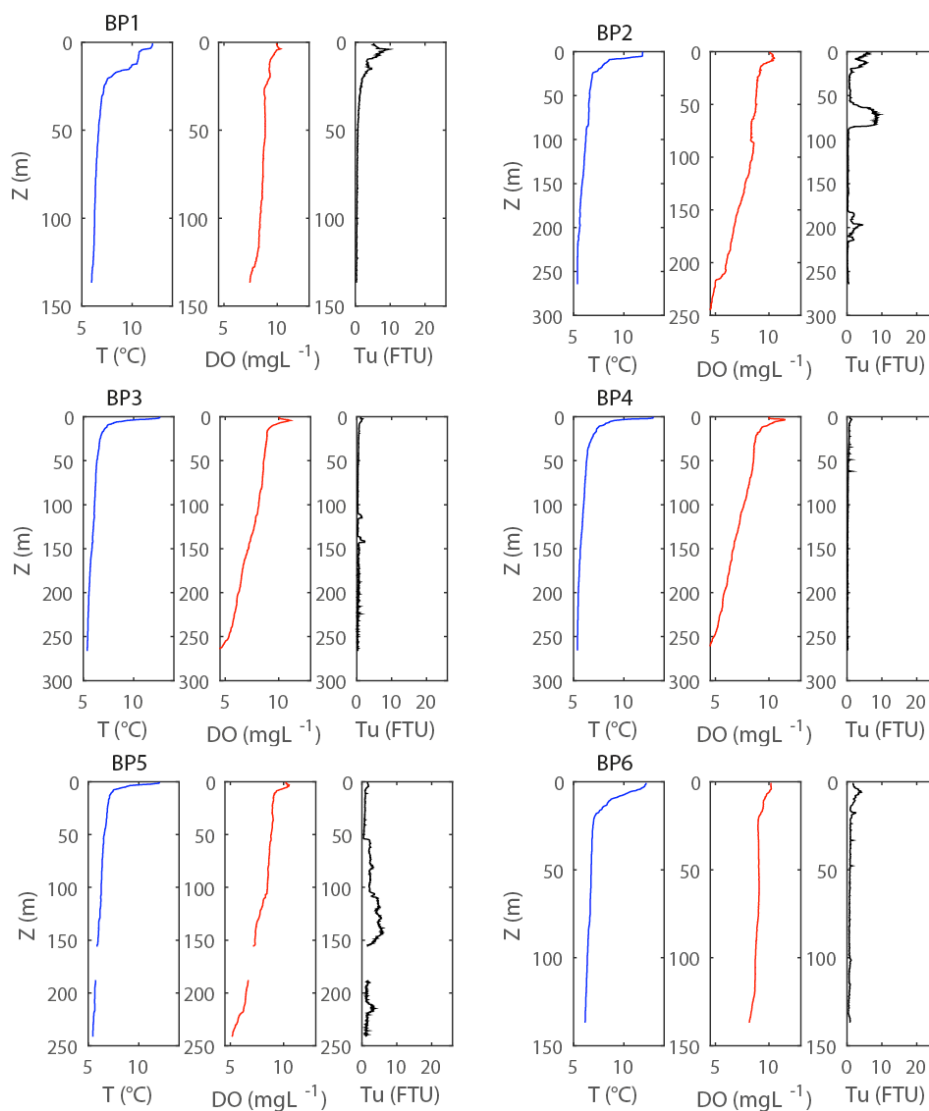
4

5

6

1

2

1

2

3

4

5

6

1

2

3

4

5

1

2   A2.

3
4

1

2   A3.

3

4

5

6

7

8

Biogeosciences
Open Access
Discussions
EGU

1

2   A4.

3

BP1	539852	148149
BP2	546865	146980
BP3	546130	145135
BP4	545300	143805
BP5	548235	143210
BP6	552530	141959
BP7	550715	141620
BP8	551700	139970
BP9	550005	140470
BP10	549005	139620
BP11	547105	140800
BP12	547745	141800
BP13	545035	140800
BP14	545565	142395
BP15	542500	143400
BP16	538714	144815
BP18	534700	144950
BP19	531138	142474
BP21	530889	140513
BP22	529829	140498
BP25	529014	140688
BP28	525772	144919
BP29	532112	146469
BP30	536069	149236

4

5