<http://www.diva-portal.org>

Postprint

This is the accepted version of a paper published in *INLAND WATERS*. This paper has been peer-reviewed but does not include the final publisher proof-corrections or journal pagination.

Citation for the original published paper (version of record):

Kosten, S., van den Berg, S., Mendonça, R., Paranaíba, J R., Roland, F. et al. (2018)
Extreme drought boosts CO₂ and CH₄ emissions from reservoir drawdown areas
INLAND WATERS, 8(3): 329-340
<https://doi.org/10.1080/20442041.2018.1483126>

Access to the published version may require subscription.

N.B. When citing this work, cite the original published paper.

Permanent link to this version:

<http://urn.kb.se/resolve?urn=urn:nbn:se:uu:diva-371238>

Extreme drought boosts CO₂ and CH₄ emissions from reservoir drawdown areas

Sarian Kosten¹, Sanne van den Berg^{1,2}, Raquel Mendonça^{3,4,#}, José R. Paranaíba⁴, Fabio Roland⁴, Sebastian Sobek³, Jamon Van Den Hoek⁵, Nathan Barros⁴

1. *Department of Aquatic Ecology and Environmental Biology, Institute for Water and Wetland Research, Radboud University Nijmegen, P.O. Box 9010 6500 GL, Nijmegen, The Netherlands*

2. *Department of Aquatic Ecology and Water Quality Management, Wageningen University, P.O. Box 47, 6708 PB, Wageningen, The Netherlands*

3. *Department of Ecology and Genetics, Limnology, Uppsala University, Uppsala, Sweden*

4. *Biology Department, Federal University of Juiz de Fora, Juiz de Fora, Brazil*

5. *Geography and Geospatial Science, College of Earth, Ocean, and Atmospheric Sciences, Oregon State University, 104 CEOAS Administration Building, 101 SW 26th St., Corvallis, OR 97331 USA*

* fm.raquel@yahoo.com.br

SK wrote the manuscript with input from all co-authors. RM, NB, FR and SvdB designed the experiment, SvdB and JR conducted the experiment, SK and SvdB analysed the experimental data, JVDH analysed the satellite images.

Extreme drought boosts CO₂ and CH₄ emissions from reservoir drawdown areas

Abstract

While previous studies suggest that greenhouse gas (GHG) emissions from reservoir sediment exposed to the atmosphere during drought may be substantial, this has not been rigorously quantified. Here, we determined carbon dioxide (CO₂) and methane (CH₄) emissions from sediment cores exposed to a drying and rewetting cycle. We found a strong temporal variation in GHG emissions with peaks when the sediment was drained (emissions from permanently wet sediment and drained sediments were respectively 251 and 1646 mg C m⁻² d⁻¹ for CO₂ and 0.8 and 547.4 mg C m⁻² d⁻¹ for CH₄) and then again during rewetting (emissions from permanently wet sediment and rewetted sediments were respectively 456 and 1725 mg C m⁻² d⁻¹ for CO₂ and 1.3 and 3.1 mg C m⁻² d⁻¹ for CH₄). To obtain insight into the importance of these emissions at a regional scale, we used Landsat satellite imagery to upscale our results to all Brazilian reservoirs. We found that during the extreme drought of 2014/2015 an additional 1299 km² of sediment was exposed, resulting in an estimated emission of 8.5x10¹¹ g CO₂-eq during the first 15 days after the overlying water disappeared and in the first 33 days after rewetting, which is of the same order of magnitude as the year-round GHG emissions of large (~mean surface water area 454 km²) Brazilian reservoirs excluding the emissions from the draw-down zone. Our estimate, however, has high uncertainty, with actual emissions likely to be higher. We therefore argue that the effects of drought on reservoir GHG emissions merits further study, especially since climate models indicate an increase in the frequency of severe droughts in the future. We recommend incorporation of emissions during drying and rewetting into GHG budgets of reservoirs to improve regional GHG emission estimates and to enable comparison between GHG emissions from hydroelectric and other electricity sources. We also highlight that in order to make reliable emission estimates, emphasis should be given to quantify peak emissions occurring at the onset of drought and the later rewetting.

Keywords: Reservoirs; sediment; drought; rewetting; emission peaks; greenhouse gases

53 **Introduction**

54 Hydroelectric reservoirs are subject to an ongoing debate regarding their net effect on
55 atmospheric greenhouse gas (GHG) concentrations (Arvizu et al. 2011) as they are
56 generally sites of intense GHG emissions (eg. Louis et al. 2000; Giles 2006; Barros et
57 al. 2011; Deemer et al. 2016) as well as strong carbon sinks due to carbon burial
58 (Mendonça et al. 2012). The debate continues largely because of the difficulties of
59 consistent and systematic monitoring of hydroelectric reservoir emissions. Therefore,
60 considerable efforts are currently being undertaken to better quantify methane (CH₄)
61 emission through the major emission pathways, which generally are ebullition from the
62 reservoir surface as well as emissions downstream from the dam (Fearnside 2015;
63 Prairie et al. 2017). A critically overlooked potential source of reservoir GHG emissions
64 however, comes from bottom sediments that are exposed to the atmosphere. While
65 previous studies suggest that GHG emissions from sediment going through a drought
66 and rewetting cycle may be substantial (eg. von Schiller et al. 2014; Gómez-Gener et al.
67 2016; Jin et al. 2016), such effects have not been rigorously quantified.

68 Reservoirs generally experience seasonal water level fluctuations, exposing
69 marginal sediments to the air. This regular exposure – in combination with sediment
70 focusing – may prevent the build-up of organic matter in these marginal sediments
71 (Mendonça et al. 2014). In areas of deeper water, however, considerable amounts of
72 organic matter can accumulate (Mendonça et al. 2014). Annual global carbon burial in
73 inland water sediments has recently been estimated to be 0.15 Pg C, of which 40% is
74 stored in reservoirs, even though these systems represent <10% of the total inland water
75 area (Mendonça et al. 2017). Organic carbon burial efficiency estimates – i.e. the
76 percentage of the carbon that remains in the sediment and is not re-mineralized – range
77 from 9% to 94%, with the highest percentages found in deepest parts of the reservoir

(Kunz et al. 2011; Sobek et al. 2012; Mendonça et al. 2016). Still, sediment accumulated at the reservoir bottom is not inert and may to some extent be remineralized, particularly when the water level drops and the sediment is exposed to air. The few existing drought studies describing carbon loss in sediments (e.g. De Groot and Van Wijck 1993; Gómez et al. 2012; Skinner et al. 2014) suggest 40% to 79% of the carbon accumulated in the top sediment layer may be lost when it dries out.

Though extended droughts are expected to occur, with higher frequency in many parts of the world (IPCC 2014), little is known about how the increasing duration and intensity of droughts will affect global surface water dynamics (Pekel et al. 2016), aquatic sediments and the regional carbon flux. Part of the carbon present in reservoir sediments will be emitted to the atmosphere as CO₂ and CH₄. Additionally, elevated release rates of dissolved organic or inorganic carbon (DOC and DIC) from the sediment to the water layer may occur after re-wetting and can lead to increased GHG emissions over an expanded spatial extent.

Although previous studies have shown that GHG emissions strongly fluctuate during a drying and rewetting cycle (e.g. Yang et al. 2012; Gallo et al. 2014; Serça et al. 2016), most assessments are based on infrequent measurements, varying from once a year along a transect (Serça et al. 2016) to twice a month at specific locations (Yang et al. 2012). As pulsed fluxes may be critical for the total GHG emissions, especially during a period of rewetting (Gallo et al. 2014), infrequent measurements are likely to result in an underestimation of overall emissions. Additionally, drawdown GHG assessments so far have focused principally on areas that experience recurring drought, while emissions may be greatest during and after a first-time drying and rewetting event (Yu et al. 2014). Here, we investigated the importance of drought and rewetting for reservoir sediment GHG emissions based on measurements of GHG emissions three-

times-a-week during an experimental drying and rewetting cycle using sediment from a single reservoir. Reservoir sediment characteristics vary greatly among (Jorcin and Nogueira 2005; Cardoso et al. 2014) and within reservoirs (Bini et al. 1999; Cardoso et al. 2013), implying there are no representative sediments to study carbon emissions during drought and re-wetting. Carbon emissions from drying sediments tend to increase with organic carbon (OC) content (Gallo et al. 2014). As the organic matter content in Brazilian reservoirs ranges from <1 to 21% (Jorcin and Nogueira 2005; Cardoso et al. 2013), we chose the oligotrophic Chapéu D'Uvas reservoir with low sediment OC content (<1.3%) to obtain a conservative estimate of potential sediment carbon emissions. To obtain insight into the importance of the extreme drought-induced GHG emissions on a regional scale, we combined our sediment carbon emissions with satellite-derived estimates of the area of sediments in Brazilian reservoirs that are generally inundated but that were exposed during Brazil's 2014/2015 extreme drought.

Materials and methods

Sediment carbon emissions

Study site

The Chapéu D'Uvas dam (21° 33'S - 43° 35'W) was constructed in the Paraibuna River - Minas Gerais, Brazil. Its catchment is situated in the Atlantic Forest biome (Ibge 2012), with a warm temperate climate (Alvares et al. 2013), characterized by warm and rainy summers and dry winter. The annual means of rainfall and temperature are 1,600 mm and 18 – 20 °C respectively (Alvares et al. 2013). The dam became operational in December 1994 and the reservoir covers 14.6 km of the Paraibuna River course, with 41 m maximum depth at the dam (mean depth: 19 m). The average surface of the reservoir is 12 km².

Sampling and analyses

The effects of drought and rewetting on sediment carbon emissions were tested using undisturbed sediment cores sampled near the littoral zone of Chapéu D'Uvas reservoir below the average low-water-line. Eight sediment cores were taken in June 2015 with a gravitational sediment corer (Uwitec) at a permanently inundated site (water depth at time of sampling was 6 m). The upper 10 cm were transferred to transparent PVC incubation cores (5.4 cm inner diameter and 41.6 cm height) without visible disturbance of the sediment. Upon arrival in the laboratory, the water above the cores was carefully removed and substituted with distilled water lacking any dissolved organic carbon (DOC), so that all the respiration measured could be attributed to the sediment. Subsequently, the cores were kept at a temperature of 27°C (+/- 3°C) with a 12-h light/12-h dark cycle under well-ventilated conditions to enhance evaporation.

Twenty-four hours after the cores were placed under these climate conditions we started our first GHG measurements. These measurements were repeated three times a week (encompassing the inundated, drying, dry and re-wetting period; see below). To measure the GHG flux, the tops of the incubation cores were closed with a gastight expandable polyvinyl chloride (PVC) stopper fitted with double O-rings, leaving at least 10 cm of headspace air between the water or sediment surface and the stopper. The stopper had an inlet and outlet port, which was connected to an Ultraportable Greenhouse Gas Analyzer (UGGA, by Los Gatos Research Inc.). We determined diffusive CO₂ and CH₄ rates based on the linear increase in concentration of the gases over a period of three minutes taking the volume of the headspace into account (as in Almeida *et al.* (2016)). When a non-linear increase was observed, caused by ebullition, we repeated the measurement. This happened only a few times. The reported carbon

emissions to the headspace are conservative, as the gases might have dissolved in the overlying water.

The water level was kept constant during the first week by adding distilled water to compensate for loss due to evaporation. In the second week, we randomly assigned four cores to a drought treatment (drought cores). From these cores, we removed 100 ml water each day until all overlying water was removed. This corresponds to a water level drop of approximately 4.3 cm d^{-1} and resembles changes in water level during dry periods observed in the reservoirs (observed water level drops in Três Marias and Mascarenhas de Moraes, two different reservoirs located in Minas Gerais, were 2.6 cm d^{-1} and 7.6 cm d^{-1} respectively). In the other four cores (permanently wet cores) the water level was kept constant. After four weeks, water levels in the drought cores were raised again to pre-drought levels by adding 100 ml distilled water every day for one week. Although we did not measure moisture content of the sediment after all overlying water was removed, we conducted a similar experiment later (sediment sampled at the same location, same cores, same incubation conditions) which pointed out that the gravimetric water content after 4 weeks of drought was approximately 28%.

After rewetting, GHG fluxes were measured for another four weeks. We distinguish four different periods: The ‘inundated period’, which refers to the period all sediments were inundated (days 0 – 11); the ‘drying period’ which refers to the period from the moment all overlaying water was removed in the drought treatment until CO_2 and CH_4 emissions in the drought cores ceased to decline and stabilized (days 11 – 26); the ‘dry period’ (days 26 – 40); and the ‘rewetting period’ from the moment when we added water to the drought cores until the end of the incubation (days 40 – 73). Arguably the exact delineation of the drying and dry period could have been chosen differently as moisture content might have further declined during the dry period.

Nevertheless, we distinguished these two periods to avoid a possible increase in emissions during the drying phase being masked by a decrease in emissions during the drought period when we analyzed the four-week period in its entirety.

Total CO₂ and CH₄ emissions during the different periods were assessed by calculating the area under the curve in the emission versus time plots. Average emissions during the different periods were calculated by dividing the total emission during the respective periods by the length of the period. CO₂ equivalents were calculated by multiplying CH₄ emissions by 32, on the basis of CH₄ global warming potential on a 100-year time horizon (Neubauer and Megonigal 2015).

In order to assess a possible DIC and DOC release from sediment to the overlying water after rewetting, at the end of the incubation the water overlying the sediment was carefully removed and subsamples were taken for total organic carbon (TOC), dissolved organic carbon (DOC) and dissolved inorganic carbon (DIC) analyses. DOC samples were filtered through a 0.6 µm pore size filter (MN GF-3, Macherey-Nagel) prior to analyses. TOC, DOC and DIC concentrations were analyzed with an infrared gas analyzer (Shimadzu, model TOC-L) at a combustion temperature of 720°C. Sediment total carbon content was determined on dried sediments (dried at 105 °C) using the Shimadzu TOC-L series, SSM50000A, in 50 mg of homogenized subsamples at 900°C. Inorganic carbon content was determined in 50 mg homogenized subsamples using the same device at 200°C and adding 0.3 mL of phosphoric acid (10 M). The difference between two values is the organic carbon in the sediment sample.

Areal extent of additional drawdown area

We estimated the areal extent of the nation-wide additional drawdown area (ADDA) during the drought of 2014/2015. This equals to the area of sediment that stayed inundated during earlier (here 2011-2013) seasonal dry periods but was exposed during

the extreme drought of 2014/2015. We used the following approach: 1) Assess the ADDA of a selected set of reservoirs based on satellite image analysis (for details see “Satellite image-driven estimates of reservoir surface area dynamics”); 2) For each of these reservoirs, determine the proportion of the ADDA with respect to maximum reservoir area as derived from Lehner et al (2011) or from our satellite image analyses (further referred to as ADDA%); 3) Assess the difference in minimum water level between the periods 2011-2013 and 2014-2015 for 137 reservoirs from the Brazilian National Agency of Water (ANA 2016); 4a) Using the reservoirs with a known percentage of ADDA (assessed in step 2) and the difference in minimum water level (assessed in step 3) we constructed a regression model with the decrease in minimum water level as the explanatory variable and the percentage of ADDA as the dependent variable; 4b) We used this regression model to estimate the ADDA of the reservoirs that were not included in the satellite image analyses.

Satellite image-driven estimates of reservoir surface area dynamics

We processed all available (970 in total) surface reflectance-corrected 30 m spatial resolution satellite images from Landsat 5 (115 images), Landsat 7 (551 images), and Landsat 8 (304 images) collected over 34 reservoirs located throughout eastern Brazil (Fig. 3) for 2011-2014. Following Van Den Hoek et al. (in prep.), surface water, land, cloud, and cloud shadow pixels were identified using the Fmask product (see http://landsat.usgs.gov/documents/provisional_lasrc_product_guide.pdf), and ‘missing’ water pixels due to cloud or shadow contamination or data gaps were addressed by ‘backfilling’ water pixels from near-date (i.e. within 32 days) images. Of these 34 reservoirs, 17 had a lower minimum water level in 2014-2015 than in the preceding years (based on the water level data of ANA (2016)). The maximum water surface area for each of these 17 reservoirs was calculated, and each reservoir’s boundary was

clipped at upstream and downstream (dam wall) locations based on visual interpretation of very high-resolution Google Maps-hosted satellite imagery. For each of the 17 reservoirs, we then determined the Landsat-derived surface area on the three dates closest to the minimum *in situ* water level date (for 2011-2013 and 2014). We visually inspected each candidate surface area and eliminated all with apparent cloud cover artefacts that reduce the surface area estimate. Of the remaining image dates, we selected those closest to the date of minimum *in situ* water level. The average temporal offset between the date of selected satellite-derived area and minimum *in situ* elevation was 22.7 days and 19.2 days for 2011-2013 and 2014, respectively. For further calculations, we only used areal estimates from within 20 days of the date of the lowest recorded water level.

Statistical analyses

Differences in total CO₂ and CH₄ emissions (area under the flux vs time curve) from the inundated and dry cores during the four different incubation periods were assessed with an unpaired t-test using SPSS (IBM).

Results

Sediment carbon emissions

Average sediment CO₂ and CH₄ emissions from the permanently wet cores over the entire incubation period were 412 (SD 139) mg C m⁻² d⁻¹ and 1.58 (SD 3.30) mg C m⁻² d⁻¹ respectively. Carbon emissions from the drought cores varied considerably over the incubation period (Table 1, Figs 1 and 2). During the drying period peak CO₂ emissions up to 6763 mg C m⁻² d⁻¹ were recorded. Total CO₂ emission during the drying period in the drought cores was 6.6 times higher than CO₂ emissions during the same time

interval in the wet cores (Table 1). CH₄ emissions reached even higher values, with maxima up to 9220 mg C m⁻² d⁻¹. During the dry period, CO₂ emissions from the drought cores were considerably lower than during the drying period. The emission rates were significantly (on average 3.0 times) lower than the emission rates from the permanently wet cores. Average CH₄ emission from the drought cores during this period was not significantly different from emission from the permanently wet cores (Table 1).

Rewetting led to an instantaneous increase in CO₂ emission, which kept increasing until approximately 9 days after rewetting. The CO₂ emissions remained higher than those from the wet cores until the end of the assay (33 days after rewetting). Peak CO₂ emissions (max. 5976 mg C m⁻² d⁻¹) were comparable to peak emissions during the drying period. Average CO₂ emissions during the rewetting period were significantly higher than emissions from the permanently wet cores (an average 3.6 fold increase). CH₄ emissions increased upon rewetting as well. CH₄ emissions from the drought cores were significantly higher (an average 2.1 fold increase) than emissions from the permanently wet cores during this rewetting period.

Upon rewetting DIC and DOC concentrations in the overlying water increased. DIC values in the overlying water differed significantly between the treatments (p<0.001) with concentrations of 5.1 mg L⁻¹ and 0.7 mg L⁻¹ in the drought and permanently wet cores, respectively. DOC concentrations at the end of the incubation were on average 49.5 mg L⁻¹ in the drought cores and 8.4 mg L⁻¹ in the permanently wet cores (which significantly differed; p<0.001), implying an additional sediment carbon loss of 12600 mg C m⁻² as a consequence of drying and rewetting.

Areal extent additional drawdown area

Images of 10 reservoirs fulfilled our requisites, i.e. we could determine their surface

water area and their ADDA within 20 days of the date of their lowest recorded water level. The largest 2014 ADDA occurred in the following reservoirs: Três Marias, Mascarenhas de Moraes and Emborcação (Table 3). The total ADDA of the 10 reservoirs was 380 km² in 2014. The maximum ADDA accounted for up to 29% of the reservoirs' surface area (Table 3). On average the ADDA comprised 10% of the reservoirs' surface area.

Forty-eight percent of the 137 reservoirs analyzed had a lower water level in 2014-2015 than in the years before. In 24% percent of the reservoirs, the minimum water level did not differ more than 10 cm between periods, and in 28% of the reservoirs the minimum water level was lower in 2011-2013 than in 2014-2015 (Fig. 3). The differences among the reservoirs are due to differences in regional precipitation patterns and river network locations. Many rivers contain several dams, leading to a cascade of reservoirs where during drought the water level of a specific reservoir can be maintained at the expense of other(s).

The relationship between ADDA% and the difference in minimum water level between the two periods studied could be described by the model:

$$\text{ADDA\%} = 7.61 \times \ln(\text{decrease in min. water level}) + 7.46$$

$$(R^2=0.78, p=0.001, n=10) \text{ (Fig. 4).}$$

As the intercept with the x-axis occurred at a minimum water level drop of 38 cm, only reservoirs with a water level drop >38 cm were included in the subsequent analysis.

We used this regression model to estimate the ADDA% and the absolute ADDA for the additional reservoirs in the ANA dataset. This led to an additional reservoir drawdown area of 919 km², and an overall estimate (satellite-derived and extrapolated) of 1299 km².

298 Discussion

299 Our drought and rewetting experiment indicates a strong impact of water level
300 fluctuation on sediment GHG emissions. We estimate that an area of 1299 km² of
301 Brazilian reservoir sediments that stayed submerged during previous dry periods were
302 exposed to the atmosphere at some point during the drought of 2014/2015. The exact
303 time of occurrence of the minimum water level varied among reservoirs with some
304 reaching their lowest levels as early as February 2014 with others as late as December
305 2015. Extrapolating the experimentally derived sediment carbon emissions to the areal
306 extent leads to an estimated emission of 3.2×10^{10} g C in the form of CO₂ and 1.1×10^{10} g
307 C in the form of CH₄ during the first 15 days of drought. Subsequent rewetting leads to
308 estimated CO₂ and CH₄ emissions of approximately 7.3×10^{10} g C in the form of CO₂
309 and 1.3×10^8 g C in the form of CH₄ during the first month after rewetting. This adds up
310 to a total estimated emission of 8.4×10^{11} g CO₂-eq at the onset of the drought and the
311 later rewetting (considering a period of 15 days of drought and 33 days of rewetting).
312 This is likely an underestimate because of several reasons, which we discuss below.

313 The intensity of the carbon emissions, i.e. the emission rate, varies strongly over
314 the drying and rewetting cycling. We observed the first peak of CO₂ and CH₄ emissions
315 when all overlying water was gone (Figs 1 and 2). As diffusion in air is > 10,000 times
316 faster than in water (Haynes and Lide 2012) the intrusion of air into the sediment upon
317 disappearance of the overlying water may strongly increase the gas exchange between
318 the sediment and the atmosphere. The resulting increase in oxygen in the sediment
319 likely increases organic matter degradation and potentially leads to high CO₂ emissions
320 as we have observed in our experiment.

321 Although we cannot substantiate that it occurred during our assay – due to a lack
322 of pore water pH and CaCO₃ measurements – peaks in CO₂ emission may also be due to

the dissolution of carbonates. When the sediment water content decreases, mineralization of organic matter may lead to a decrease in pore water pH, triggering dissolution (Skinner et al. 2014). When the water content in the sediment further decreases, microbial activity declines (Jin et al. 2016) resulting in reduced GHG emissions as observed in our experiment (Figs 1 and 2) as well as *in situ* in dry streambeds (Gómez-Gener et al. 2015).

Although methanogenesis has been found to occur under oxic conditions (Bogard et al. 2014; Tang et al. 2016) an increase in oxygen availability in drying sediment tends to decrease rather than increase CH₄ production (Segers 1998; Koschorreck 2000). Nonetheless, methanogenesis may persevere in anoxic micro sites in the upper – largely oxic – sediment layers (Jørgensen and Revsbech 1985) as well as in deeper – still anoxic – sediment layers. At the same time, CH₄ oxidation is likely to increase due to the increase in oxygen availability in the sediment (Koschorreck 2000). During the initial drying phase, CH₄ production may still be higher than CH₄ oxidation, resulting in a net-CH₄ emission (Koschorreck 2000).

In addition, an increase in CH₄ emission may occur initially when gases that were stored in deeper sediment layers are vented. Gas venting from drying sediments may be an important pathway, especially for CH₄ which, due to its low solubility in water, forms gas bubbles in the sediment. During gas venting, an air-filled pathway (consisting of pores or cracks) connects gas bubbles to the sediment surface, the trapped gas may move upwards first by convection and, when the pressure is equilibrated, move further upward by molecular diffusion (Rosenberry et al. 2006). The processes may differ from ebullition, i.e. the release of gas bubbles from the sediment and their rise through the water column, with respect to the time span during which the gas bubble reaches the atmosphere. Venting is a more gradual process as it depends partially on

diffusion whereas ebullition events lead to short-term peaks in emission when the bubbles reach the water's surface. On a daily time frame, however, the CH₄ peaks we observed during the drying phase were similar to reported ebullitive fluxes in different water systems (Aben et al. 2017). The exact timing of a venting event will vary between sites, which likely explains the large variation in the peak emissions we observed. This calls for frequent measurements of GHG emission during the drying period, to ensure accurate estimates.

Our assay shows fairly constant GHG emissions during the dry period with low sediment CH₄ and CO₂ emissions compared with permanently wet sediments (Table 1). The low emissions are likely due to desiccation of the microbial community (Borken and Matzner 2009).

Rewetting increased CH₄ emissions (Table 1). This finding corroborates numerous other studies – e.g. *in situ* measurements in Amazonian lakes by Conrad et al. (2014); and laboratory incubations by Kannenberg et al. (2015), but also see Gallo et al. (2014), who found no significant increase after re-wetting (see Table 2) which is likely linked to the re-establishment of anoxic conditions in the sediment and the switch back to anaerobic metabolism. Concurrently, rewetting has been shown to lead to an increase in microbial biomass and a shift in community composition (eg. Evans and Wallenstein 2012; Conrad et al. 2014). In our assay, CH₄ emission during the rewetting phase did not seem to differ from that during the earlier inundated phase (Table 1). Nevertheless, methanogenesis rates after rewetting have been found to exceed those of the pre-drought situation in a previous study (Kannenberg et al. 2015). The authors of that study argued that rewetting may release nutrients from rewetted sediments due to microbial lysis and disruption of soil aggregates which liberates different compounds that stimulate the microbial community and their mineralization activity. However, physical

drivers of gas emissions play a role as well. Instantaneous CO₂ emissions upon rewetting of up to 38 g C m⁻² d⁻¹ have been reported, which are argued to be indicative of displacement of gases accumulated in the pores during the dry period (Gallo et al. 2014). The peak emissions upon rewetting again call for measurements during reservoir refilling.

We likely underestimated the emissions that actually occurred during the 2014/2015 drought because of several reasons. Firstly, we likely underestimated the total areal extent of the sediments exposed to the atmosphere due to the severe 2014/2015 drought because we only included reservoirs in the ANA database with daily water level data. Secondly, our sediment-core-based estimation of GHG emission intensity due to drought and rewetting is likely to be modest. Specifically, ebullition was not captured by our measurements, yet it may contribute substantially to the total flux (Aben et al. 2017) when the sediments are inundated, especially when the water level drops (Maeck et al. 2014; Harrison et al. 2017). Although we rarely observed bubbles, they may have occurred, leading to an underestimation of the CH₄ emission. Additionally, our estimate does not include the 11.4 g C m⁻² of carbon that was emitted from the sediment in the form of DOC after rewetting. In the water column, DOC may be partially mineralized through photo-oxidation, thereby enhancing reservoir carbon emissions (Peter et al. 2016). Our experiment was conducted indoors at considerably lower light intensities than outside and hence we likely underestimate DOC mineralization by photo-oxidation. Additionally, we also may have underestimated possible CO₂ production by photo-oxidation of the dry sediment (Rutledge et al. 2010).

Arguably, the most important factor pointing to a strong underestimation of the actual fluxes is the fact that our estimates are based on measurements conducted on sediments containing little organic matter. Besides nitrogen content, sediment particle

size and temperature, sediment GHG emissions are known to depend on organic matter content and reactivity (Gudas et al. 2010; Gallo et al. 2014; Gómez-Gener et al. 2016). A comparison between sediments with organic matter contents of 3.0% and ~5.4% revealed a 1.7-fold increase in CO₂ emissions upon rewetting in the more organic sediment (Gallo et al. 2014). The sediments we used for our estimate had an organic matter content of 1.3% whereas organic matter contents in a range of Brazilian reservoirs have been found to be up to a factor 10 times higher (Bini et al. 1999; Jorcin and Nogueira 2005; Mariani and Pompêo 2008; Cardoso et al. 2013). In addition, redistribution of sediment and organic matter can lead to strong gradients in sediment organic matter content within an individual reservoir (Cardoso et al. 2013; Mendonça et al. 2016). The exposed sediment will therefore vary in organic matter content and likely in GHG emissions. This has to be taken into account in future work when comparing across reservoirs with different levels of drawdown and bathymetry. At this point, we can only speculate on how much higher the drought-induced carbon emissions from these more organic rich sediments may be, but we expect the increase will be considerable. Comparisons of our emission intensities with those reported in the few studies on GHG emissions from dry sediments confirm this: our dry sediment emissions are considerably lower than earlier reports (Table 2). Still, our dry sediment cores emitted CH₄, whereas some dry sediments may consume CH₄ (Koschorreck 2000; Juutinen et al. 2001; Yang et al. 2012; Jin et al. 2016). Additionally, the rewetting induced peak emissions may depend on the duration and severity of the dry period. When sediments do not dry out completely and aerobic mineralization continues during the dry period, organic matter availability may be limiting upon rewetting, which may result in an increase in the CO₂:CH₄ emission ratio over the whole drying and rewetting cycle. In short there are many different variables that influence GHG emissions during a

drying and rewetting cycle and we therefore highlight that the uncertainty regarding the GHG emissions due to drought and rewetting is high and our estimate is but a first step in quantifying the effect of extreme drought on Brazilian reservoir GHG emissions.

Even without taking the likely underestimation of our emission estimate into account, our analyses show that the single extreme drought event in 2014/2015 and the subsequent rewetting led to a higher total GHG emission than during an entire year by large Brazilian reservoirs (based on a comparison with the eight reservoirs (mean surface water area 454 km²) described by Ometto et al. (2013)). In one of the reservoirs (Mascarenhas de Moraes) GHG emission was estimated to be 4.1×10^{10} g CO₂-eq in the drawdown area during the first 15 days of drought and the first 33 days after rewetting. This is a 33% addition to the 1.25×10^{11} g CO₂-eq emitted from the reservoir water surface area on a yearly basis (Ometto et al. 2013).

Conclusion

Our combined estimate of the Brazilian reservoir drawdown area during the 2014-2015 drought and the effect of drying and rewetting on GHG emissions from reservoir sediment highlights the potential importance of drought and rewetting cycles for reservoir GHG-budgets. It also indicates that interpreting sediment carbon burial as a long-term carbon sink (Mendonça et al. 2012) should be done with care as disturbances such as drought may liberate large parts of this carbon stock. This applies to sediment desiccation after the decommissioning of dams as well. The number of dams that will be decommissioned will increase over the years to come and our findings are indicative of the consequent potential carbon emissions. We argue, contributing to the current IPCC discussion, that the effect of drought events on sediment GHG emissions should be incorporated in hydroelectric reservoir GHG budgets to enable a valid comparison with the greenhouse gas emissions of other electricity sources. Furthermore, as inland waters

are often hotspots of carbon emissions in terrestrial landscapes (Kosten et al. 2010; Raymond et al. 2013) substantial improvement of regional GHG emission estimates may be made when drought-induced sediment carbon emissions are taken into account. This is particularly important as severe droughts are likely to occur more frequently in the future. More work is needed to estimate these emissions accurately with special attention to peak emissions occurring at the onset of drought and at the time of rewetting as our results suggest that these peaks make up the largest part of the total emission.

Acknowledgements

We thank David Hamilton and two anonymous reviewers for their constructive comments on earlier versions of the manuscript. SK was supported by Nederlandse Organisatie voor Wetenschappelijk Onderzoek (NWO) Veni Grant 86312012. SvdB was supported by Stichting Nijmeegs Universiteitsfonds (SNUF). The research leading to these results has received additional funding from the European Research Council under the European Union's Seventh Framework Programme (FP7/2007-2013) / ERC grant agreement n° 336642." Programme (FP7/2007-2013) / ERC grant agreement n° 336642 to SS and RM, and from Fundação de Amparo à Pesquisa de Minas Gerais/FAPEMIG (CRA APQ 03045/16) to NB. FR is partially supported by CNPq.

References

Aben RCH, Barros N, van Donk E, Frenken T, Hilt S, Kazanjian G, Lamers LPM, Peeters ETHM, Roelofs JGM, de Senerpont Domis LN, Stephan S, Velthuis M, Van de Waal DB, Wik M, Thornton BF, Wilkinson J, DelSontro T, Kosten S. 2017. Cross continental increase in methane ebullition under climate change. *Nature communications*. 8(1):1682.
Almeida RM, Nóbrega GN, Junger PC, Figueiredo AV, Andrade AS, Moura CGB, Tonetta D, Oliveira Jr. ES, Araújo F, Rust F, Piñeiro-Guerra JM, Mendonça Jr. JR, Medeiros LR, Silva LP, Miranda M, Costa MRA, Melo ML, Nobre R, Benevides T,

476 Roland F, de Klein J, Barros NO, Mendonça R, Becker V, Huszar V, Kosten S. 2016.
 477 High primary production contrasts with intense carbon emission in a eutrophic tropical
 478 reservoir. *Frontiers in microbiology*. 7. English.
 479 Alvares CA, Stape JL, Sentelhas PC, de Moraes G, Leonardo J, Sparovek G. 2013.
 480 Köppen's climate classification map for Brazil. *Meteorologische Zeitschrift*. 22(6):711-
 481 728.
 482 ANA. 2016. Agência Nacional de Águas, Sistema de Acompanhamento de
 483 Reservatórios [accessed 2016 13/9/2016]. <http://sar.ana.gov.br/>.
 484 Arvizu D, Bruckner T, Chum H, Edenhofer O, Estefen S, Faaij A, Fishedick M,
 485 Hansen G, Hiriart G, Hohmeyer O. 2011. Technical summary. IPCC special report on
 486 renewable energy sources and climate change mitigation.
 487 Barros N, Cole JJ, Tranvik LJ, Prairie YT, Bastviken D, Huszar VLM, del Giorgio P,
 488 Roland F. 2011. Carbon emission from hydroelectric reservoirs linked to reservoir age
 489 and latitude. *Nature Geosci*. 4(9):593-596.
 490 Bini LM, Thomaz SM, Murphy KJ, Camargo AF. 1999. Aquatic macrophyte
 491 distribution in relation to water and sediment conditions in the Itaipu Reservoir, Brazil.
 492 *Biology, Ecology and Management of Aquatic Plants*. Springer; p. 147-154.
 493 Bogard MJ, Del Giorgio PA, Boutet L, Chaves MCG, Prairie YT, Merante A, Derry
 494 AM. 2014. Oxic water column methanogenesis as a major component of aquatic CH₄
 495 fluxes. *Nature communications*. 5:5350.
 496 Borken W, Matzner E. 2009. Reappraisal of drying and wetting effects on C and N
 497 mineralization and fluxes in soils. *Global Change Biology*. 15(4):808-824.
 498 Cardoso SJ, Enrich-Prast A, Pace ML, Roland F. 2014. Do models of organic carbon
 499 mineralization extrapolate to warmer tropical sediments? *Limnology and*
 500 *Oceanography*. 59(1):48-54.
 501 Cardoso SJ, Vidal LO, Mendonça RF, Tranvik LJ, Sobek S, Fábio R. 2013. Spatial
 502 variation of sediment mineralization supports differential CO₂ emissions from a tropical
 503 hydroelectric reservoir. *Frontiers in microbiology*. 4(101).
 504 Conrad R, Ji Y, Noll M, Klose M, Claus P, Enrich-Prast A. 2014. Response of the
 505 methanogenic microbial communities in Amazonian oxbow lake sediments to
 506 desiccation stress. *Environmental Microbiology*. 16(6):1682-1694.
 507 De Groot C-J, Van Wijk C. 1993. The impact of desiccation of a freshwater marsh
 508 (Garcines Nord, Camargue, France) on sediment-water-vegetation interactions.
 509 *Hydrobiologia*. 252(1):83-94.
 510 Deemer BR, Harrison JA, Li S, Beaulieu JJ, DelSontro T, Barros N, Bezerra-Neto JF,
 511 Powers SM, dos Santos MA, Vonk JA. 2016. Greenhouse Gas Emissions from
 512 Reservoir Water Surfaces: A New Global Synthesis. *BioScience*. 66(11):949-964.
 513 Evans SE, Wallenstein MD. 2012. Soil microbial community response to drying and
 514 rewetting stress: does historical precipitation regime matter? *Biogeochemistry*.
 515 109(1):101-116.
 516 Fearnside PM. 2015. Emissions from tropical hydropower and the IPCC. *Environmental*
 517 *Science & Policy*. 50(Supplement C):225-239.
 518 Gallo EL, Lohse KA, Ferlin CM, Meixner T, Brooks PD. 2014. Physical and biological
 519 controls on trace gas fluxes in semi-arid urban ephemeral waterways. *Biogeochemistry*.
 520 121(1):189-207.
 521 Giles J. 2006. Methane quashes green credentials of hydropower. *Nature*.
 522 444(7119):524-525.
 523 Gómez-Gener L, Obrador B, Marcé R, Acuña V, Catalán N, Casas-Ruiz JP, Sabater S,
 524 Muñoz I, Schiller D. 2016. When Water Vanishes: Magnitude and Regulation of
 525 Carbon Dioxide Emissions from Dry Temporary Streams. *Ecosystems*. 1-14.

526 Gómez-Gener L, Obrador B, Schiller D, Marcé R, Casas-Ruiz JP, Proia L, Acuña V,
 527 Catalán N, Muñoz I, Koschorreck M. 2015. Hot spots for carbon emissions from
 528 Mediterranean fluvial networks during summer drought. *Biogeochemistry*. 125(3):409-
 529 426.
 530 Gómez R, Arce M, Sánchez J, del Mar Sánchez-Montoya M. 2012. The effects of
 531 drying on sediment nitrogen content in a Mediterranean intermittent stream: a
 532 microcosms study. *Hydrobiologia*. 679(1):43-59.
 533 Gudas C, Bastviken D, Steger K, Premke K, Sobek S, Tranvik LJ. 2010. Temperature-
 534 controlled organic carbon mineralization in lake sediments. *Nature*. 466(7305):478-481.
 535 Harrison JA, Deemer BR, Birchfield MK, O'Malley MT. 2017. Reservoir water-Level
 536 drawdowns accelerate and amplify methane emission. *Environmental Science &*
 537 *Technology*. 51(3):1267-1277.
 538 Haynes WM, Lide D. 2012. *CRC Handbook of Chemistry and Physics*, CRC Handbook
 539 of Chemistry and Physics. Taylor & Francis.
 540 Ibge DdP. 2012. Coordenação de Agropecuária. Produção Pecuária Municipal. 43:1-55.
 541 Jin H, Yoon TK, Lee S-H, Kang H, Im J, Park J-H. 2016. Enhanced greenhouse gas
 542 emission from exposed sediments along a hydroelectric reservoir during an extreme
 543 drought event. *Environmental Research Letters*. 11(12):124003.
 544 Jorcin A, Nogueira MG. 2005. Temporal and spatial patterns based on sediment and
 545 sediment–water interface characteristics along a cascade of reservoirs (Paranapanema
 546 River, south-east Brazil). *Lakes & Reservoirs: Research & Management*. 10(1):1-12.
 547 Jørgensen BB, Revsbech NP. 1985. Diffusive boundary layers and the oxygen uptake of
 548 sediments and detritus. *Limnology and Oceanography*. 30(1):111-122.
 549 Juutinen, Alm, Martikainen, Silvola. 2001. Effects of spring flood and water level draw-
 550 down on methane dynamics in the littoral zone of boreal lakes. *Freshwater Biology*.
 551 46(7):855-869.
 552 Kannenberg SA, Dunn ST, Ludwig SM, Spawn SA, Schade JD. 2015. Patterns of
 553 Potential Methanogenesis Along Soil Moisture Gradients Following Drying and
 554 Rewetting in Midwestern Prairie Pothole Wetlands. *Wetlands*. 35(4):633-640.
 555 Koschorreck M. 2000. Methane turnover in exposed sediments of an Amazon
 556 floodplain lake. *Biogeochemistry*. 50(2):195-206.
 557 Kosten S, Roland F, Da Motta Marques DML, Van Nes EH, Mazzeo N, Sternberg
 558 LdSL, Scheffer M, Cole JJ. 2010. Climate-dependent CO₂ emissions from lakes. *Global*
 559 *Biogeochemical Cycles*. 24(2):GB2007.
 560 Kunz MJ, Anselmetti FS, Wüest A, Wehrli B, Vollenweider A, Thüning S, Senn DB.
 561 2011. Sediment accumulation and carbon, nitrogen, and phosphorus deposition in the
 562 large tropical reservoir Lake Kariba (Zambia/Zimbabwe). *Journal of Geophysical*
 563 *Research: Biogeosciences*. 116(G3):n/a-n/a.
 564 Lehner B, Liermann CR, Revenga C, Vörösmarty C, Fekete B, Crouzet P, Döll P,
 565 Endejan M, Frenken K, Magome J. 2011. High-resolution mapping of the world's
 566 reservoirs and dams for sustainable river-flow management. *Frontiers in Ecology and*
 567 *the Environment*. 9(9):494-502.
 568 Louis VLS, Kelly CA, Duchemin É, Rudd JW, Rosenberg DM. 2000. Reservoir
 569 Surfaces as Sources of Greenhouse Gases to the Atmosphere: A Global Estimate
 570 Reservoirs are sources of greenhouse gases to the atmosphere, and their surface areas
 571 have increased to the point where they should be included in global inventories of
 572 anthropogenic emissions of greenhouse gases. *BioScience*. 50(9):766-775.
 573 Maeck A, Hofmann H, Lorke A. 2014. Pumping methane out of aquatic sediments:
 574 ebullition forcing mechanisms in an impounded river. *Biogeosciences*. 11(11):2925-
 575 2938.

576 Mariani CF, Pompêo ML. 2008. Potentially bioavailable metals in sediment from a
577 tropical polymictic environment—Rio Grande Reservoir, Brazil. *Journal of Soils and*
578 *Sediments*. 8(5):284.

579 Mendonça R, Kosten S, Sobek S, Barros N, Cole JJ, Tranvik L, Roland F. 2012.
580 Hydroelectric carbon sequestration. *Nature Geoscience*. 5(12):838-840.

581 Mendonça R, Kosten S, Sobek S, Cardoso SJ, Figueiredo-Barros MP, Estrada CHD,
582 Roland F. 2016. Organic carbon burial efficiency in a subtropical hydroelectric
583 reservoir. *Biogeosciences*. 13(11):3331-3342.

584 Mendonça R, Kosten S, Sobek S, Cole J, Bastos A, Albuquerque A, Cardoso S, Roland
585 F. 2014. Carbon Sequestration in a Large Hydroelectric Reservoir: An Integrative
586 Seismic Approach. *Ecosystems*. 17(3):430-441. English.

587 Mendonça R, Müller RA, Clow D, Verpoorter C, Raymond P, Tranvik LJ, Sobek S.
588 2017. Organic carbon burial in global lakes and reservoirs. *Nature communications*.
589 8(1):1694.

590 Neubauer SC, Megonigal JP. 2015. Moving beyond global warming potentials to
591 quantify the climatic role of ecosystems. *Ecosystems*. 18(6):1000-1013.

592 Ometto JP, Cimbleris ACP, dos Santos MA, Rosa LP, Abe D, Tundisi JG, Stech JL,
593 Barros N, Roland F. 2013. Carbon emission as a function of energy generation in
594 hydroelectric reservoirs in Brazilian dry tropical biome. *Energy Policy*. 58:109-116.

595 Pekel J-F, Cottam A, Gorelick N, Belward AS. 2016. High-resolution mapping of
596 global surface water and its long-term changes. *Nature*. 540(7633):418-422.

597 Peter S, Isidorova A, Sobek S. 2016. Enhanced carbon loss from anoxic lake sediment
598 through diffusion of dissolved organic carbon. *Journal of Geophysical Research:*
599 *Biogeosciences*. 121(7):1959-1977.

600 Prairie YT, Alm J, Beaulieu J, Barros N, Battin T, Cole J, del Giorgio P, DelSontro T,
601 Guérin F, Harby A, Harrison J, Mercier-Blais S, Serça D, Sobek S, Vachon D. 2017.
602 Greenhouse Gas Emissions from Freshwater Reservoirs: What Does the Atmosphere
603 See? *Ecosystems*.

604 Raymond PA, Hartmann J, Lauerwald R, Sobek S, McDonald C, Hoover M, Butman D,
605 Striegl R, Mayorga E, Humborg C, Kortelainen P, Durr H, Meybeck M, Ciais P, Guth
606 P. 2013. Global carbon dioxide emissions from inland waters. *Nature*. 503(7476):355-
607 359.

608 Rosenberry DO, Glaser PH, Siegel DI. 2006. The hydrology of northern peatlands as
609 affected by biogenic gas: current developments and research needs. *Hydrological*
610 *Processes*. 20(17):3601-3610.

611 Rutledge S, Campbell DI, Baldocchi D, Schipper LA. 2010. Photodegradation leads to
612 increased carbon dioxide losses from terrestrial organic matter. *Global Change Biology*.
613 16(11):3065-3074.

614 Segers R. 1998. Methane production and methane consumption: a review of processes
615 underlying wetland methane fluxes. *Biogeochemistry*. 41(1):23-51.

616 Serça D, Deshmukh C, Pighini S, Oudone P, Vongkhamsoo A, Guédant P, Rode W,
617 Godon A, Chanudet V, Descoux S. 2016. Nam Theun 2 Reservoir four years after
618 commissioning: significance of drawdown methane emissions and other pathways.
619 *Hydroécologie Appliquée*. 19:119-146.

620 Skinner D, Oliver R, Aldridge K, Brookes J. 2014. Extreme water level decline effects
621 sediment distribution and composition in Lake Alexandrina, South Australia.
622 *Limnology*. 15(2):117-126.

623 Sobek S, DelSontro T, Wongfun N, Wehrli B. 2012. Extreme organic carbon burial
624 fuels intense methane bubbling in a temperate reservoir. *Geophysical Research Letters*.
625 39(1):n/a-n/a.

626 Tang KW, McGinnis DF, Ionescu D, Grossart H-P. 2016. Methane Production in Oxic
627 Lake Waters Potentially Increases Aquatic Methane Flux to Air. *Environmental Science*
628 *& Technology Letters*. 3(6):227-233.
629 von Schiller D, Marcé R, Obrador B, Gómez L, Casas JP, Acuña V, Koschorreck M.
630 2014. Carbon dioxide emissions from dry watercourses. *Inland waters*. 4(4):377-382.
631 Yang L, Lu F, Wang X, Duan X, Song W, Sun B, Chen S, Zhang Q, Hou P, Zheng F.
632 2012. Surface methane emissions from different land use types during various water
633 levels in three major drawdown areas of the Three Gorges Reservoir. *Journal of*
634 *Geophysical Research: Atmospheres*. 117(D10).
635 Yu Z, Wang G, Marschner P. 2014. Drying and rewetting – Effect of frequency of
636 cycles and length of moist period on soil respiration and microbial biomass. *European*
637 *Journal of Soil Biology*. 62:132-137.

638

639

Table 1. Average and total sediment CO₂ and CH₄ emissions from permanently wet sediments (“wet”) and sediments that were subjected to drought and subsequent rewetting (“drought”; SD in between parentheses, n=4). p-value represents significant difference between treatments based on t-test. Significant differences are presented in bold.

	CO ₂ emissions					CH ₄ emissions				
	Flux		Total emission			Flux		Total emission		
	[mg C m ⁻² d ⁻¹]		[mg C m ⁻²]			[mg C m ⁻² d ⁻¹]		[mg C m ⁻²]		
	Wet	Drought	Wet	Drought	p	Wet	Drought	Wet	Drought	p
Inundated period	443 (35)	402 (56)	5010 (392)	4546 (632)	0.266	1.9 (0.9)	8.1 (11.0)	21.8 (10.3)	91.9 (124.0)	0.303
Drying period	251 (82)	1646 (392)	3718 (1220)	24356 (5797)	<0.0001	0.8 (0.9)	547.4 (688.8)	12.1 (13.8)	8102.2 (10194)	0.164
Dry period	449 (99)	148 (147)	6335 (1394)	2091 (2078)	0.018	2.8 (3.1)	2.2 (0.5)	39.5 (44.3)	30.6 (6.5)	0.705
Re-wetting period	456 (62)	1725 (181)	14973 (2035)	56569 (5941)	<0.0001	1.3 (0.3)	3.1 (0.3)	41.8 (10.9)	100.8 (10.8)	<0.0001
Total period			30037 (2625)	87561 (8899)	<0.0001			115.2 (48.6)	8325.5 (10324.1)	0.210

Table 2: CO₂ and CH₄ emissions from temporarily exposed and inundated sediments in different water systems

Location	System type	Measurement frequency	Condition	Mean flux (mg C m ⁻² d ⁻¹)		Reference
				CO ₂ -C	CH ₄ -C	
Brazil	river	7-9 times over a period of 2 months in 1996 and 1997	Initial phase of dry period		11.5 ±8.6	Kosschoreck (2000)
			After approx. 2 months		-0.84 ±0.34	
Finland	lake	1 or 3 times a week along transects for 4 to 6 months during water level lowering	Dry - permanently wet with emergent plants		-3.6 - 255.6	Juutinen et al. (2001)
China	reservoir	once or twice a month in fallow, crop and deforested land in drawdown zone	Dry		-0.144 ±0.63 (SE)	Yang et al. (2012)
			Wet (before and after drought)		3.96 ± 4.68 (SE)	
Arizona, USA	urban stream	1 hour before rewetting, immediately after and 0.5, 2 and 6 hours after rewetting	Dry	526 ± 101 (SE)	0.41±0.12 (SE)	Gallo et al. (2014)
			Re-wetted	5760 ± 816 (SE)	not diff. from dry	
			Instantaneous after re-wetting	10013 ±2530 (SE)	not diff. from dry	
Spain	temporary stream	2 times, once during dry period, once during wet	Dry	9377 ±4682(SD)		Gomez et al. (2016)
			Flowing streambed	3667 ±2473(SD)		
Laos	reservoir	transect in 2010 and 2011 in drawdown zone	Dry (soil moisture 8-40%)		0.72± 2.4	Serça et al. (2016)
			Wet (soil moisture>40%)		276±348	
South Korea	reservoir	once along a transect	50 m from waterline	12000 ± 2500 (SE)		Jin et al. (2016)
			140 m from waterline	3000 ± 900 (SE)		

Table 3. Minimum water surface area (derived from Landsat images) of 17 Brazilian reservoirs with lower minimum water levels in 2014 than during the 3 preceding years. For the 11 reservoirs for which we had areal estimates within a time span of 20 days from the date (mm-dd-yy) the lowest water level was recorded, we also calculated the additional draw down area (ADDA) and the percentage of ADDA of the total reservoir area.

reservoir/dam name	2011-2013			2014			comparison between periods		
reservoir name	date of lowest water level	min. water surface area [km ²]	time span [days]	date of lowest water level	min. water surface area [km ²]	time span [days]	difference in min. water level [m]	additional draw down area [km ²]	ADDA of total reservoir area [%]
Três Marias	26-11-2013	494	6	14-11-2014	353	7	6.30	140	18
Mascarenhas de Moraes	3-1-2013	212	16	11-12-2014	150	5	8.14	63	25
Emborcação	1-1-2011	215	12	27-11-2014	154	6	5.67	61	29
Sao Simão	11-11-2012	516	13	28-10-2014	479	17	0.96	38	6
Itaipú	27-12-2013	1191	2	11-2-2014	1160	0	0.62	32	3
Barra Bonita	28-12-2011	213	12	12-12-2014	191	4	0.92	23	11
Chavantes	2-2-2013	307	9	19-12-2014	298	4	0.85	9	3
Serra de Mesa	16-12-2013	799	20	12-12-2014	790	13	1.252	9	1
Marimbondo	13-12-2012	196	20	27-11-2014	192	10	0.19	4	1
Nova Avanhandava	5-12-2013	173	2	8-5-2014	173	4	0.54	1	0.3
Sobradinho	4-12-2013	2465	35	4-12-2014	1990	34	0.92		
Ilha Solteira	2-1-2013	1124	65	17-11-2014	962	61	6.18		
Furnas	14-12-2012	1000	67	13-11-2014	825	38	0.21		
Promissão	24-12-2011	531	33	26-2-2014	529	20	0.43		
Capivara	14-12-2012	490	62	13-12-2014	492	47	1.42		
Nova Ponte	11-1-2013	224	8	27-11-2014	196	37	8.87		
Estreito (L.C.B. de Carvalho)	4-9-2013	60	4	9-4-2014*	44	19	0.21		

* This image and the subsequent image taken on 17-4-2014 had high cloud contamination and were therefore not included in the ADDA analysis.

Figure 1. CO₂ (a) and CH₄ (b) emissions (average and SEM) from sediments that were continuously inundated (wet cores) and sediments that were exposed to drying and subsequent rewetting (drought cores).

Figure 2. Difference between the CO₂ and CH₄ emissions from sediments that were exposed to drying and subsequent rewetting (drought cores) and from sediments that remained wet for the entire incubation period (permanently wet cores). Positive values (surplus) signify higher emissions from the drought cores than from the wet cores.

Figure 3. Comparison of minimum water levels in 137 reservoirs. Black circles signify reservoirs with satellite-derived estimates of the 2014 additional drawdown area.

Figure 4. Relation between the percentage of additional drawdown area (ADDA%) and the decrease in minimum water level between the periods 2011-2013 and 2014-2015.

The dotted line depicts the regression line $ADDA\% = 7.61 \times \ln(\text{decrease in min. water level}) + 7.46$