Impacts of Extreme Flooding on Riverbank Filtration Water Quality

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Abstract

Riverbank filtration schemes form a significant component of public water treatment processes on a global level. Understanding the resilience and water quality recovery of these systems following severe flooding is critical for effective water resources management under potential future climate change. This paper assesses the impact of floodplain inundation on the water quality of a shallow aquifer riverbank filtration system and how water quality recovers following an extreme (1 in 17 year, duration > 70 days, 7 day inundation) flood event. During the inundation event, riverbank filtrate water quality is dominated by rapid direct recharge and floodwater infiltration (high fraction of surface water, dissolved organic carbon (DOC) > 140% baseline values, > 1 log increase in micro-organic contaminants, microbial detects and turbidity, low specific electrical conductivity (SEC) < 90% baseline, high dissolved oxygen (DO) > 400% baseline). A rapid recovery is observed in water quality with most floodwater impacts only observed for 2 - 3 weeks after the flooding event and a

return to normal groundwater conditions within 6 weeks (lower fraction of surface water, higher SEC, lower DOC, organic and microbial detects, DO). Recovery rates are constrained by the hydrogeological site setting, the abstraction regime and the water quality trends at site boundary conditions. In this case, increased abstraction rates and a high transmissivity aquifer facilitate rapid water quality recoveries, with longer term trends controlled by background river and groundwater qualities. Temporary reductions in abstraction rates appear to slow water quality recoveries. Flexible operating regimes such as the one implemented at this study site are likely to be required if shallow aquifer riverbank filtration systems are to be resilient to future inundation events. Development of a conceptual understanding of hydrochemical boundaries and site hydrogeology through monitoring is required to assess the suitability of a prospective riverbank filtration site.

Keywords

Riverbank filtration, flooding, hydrochemistry, water supply management

1. Introduction

Riverbank filtration (RBF) is a primary water treatment methodology where river water infiltrates through an alluvial aquifer to collector wells. Water derived from collector wells is generally cleaner than that extracted from the river directly (Eckert and Irmscher, 2006) and can reduce further treatment costs. RBF systems are commonplace for public water supply in many countries. In Europe, riverbank filtration systems have been in place since 1870 (Schubert, 2002). Infiltrating river water provides 50% of the public water supply of Slovakia, 45% in Hungary and 16% in Germany (Hiscock and Grischek, 2002). In the United States, riverbank filtration systems have been used for more than 50 years (Ray et al., 2002a). Figure 1 (a) shows the spatial distribution of riverbank filtration sites in England. Using environmental regulator abstraction licence data (Environment Agency, 2014) in conjunction with alluvial aquifer and river mapping, we estimate that shallow groundwater systems with a component of riverbank filtration supply approximately 900 Ml/day. This corresponds to approximately 10% of total annual licenced groundwater supply. Grooters (2006) showed that riverbank filtration reduced costs of reverse-osmosis treatment of surface waters by 10 – 20%.

RBF systems exploit the natural physical, biological and chemical processes which occur between the river and the collector well to reduce contaminant loadings (Hiscock and Grischek, 2002). Changes in water quality occurring from the river through the hyporheic zone to the collector well have been well characterised. Along this pathway it is considered that there are two distinct biogeochemical zones with different attenuation processes occurring. A biologically active colmation (clogging) layer is present below the river bed where intensive degradation and sorption can occur. The flow path to the collector well has less capacity for sorption and degradation but reduced contaminant concentrations through mixing and dilution is common. Numerous studies have shown riverbank filtration to be effective in removal and/or degradation of microorganisms, turbidity, pesticides, dissolved and total organic carbon and organic micropollutants (Weiss et al. (2005); Dash et al. (2010); Verstraeten et al. (2002), Grünheid et al. (2005), Maeng et al. (2010), Hoppe-Jones et al. (2010), Hiscock and Grischek (2002) and references therein).

RBF systems are considered to be vulnerable to climate change (Sprenger et al., 2011). Increased frequency and severity of extreme floods and droughts under climate change has the potential to affect both riverbank filtrate water quality and quantity. Using a hypothetical flooding scenario, Sprenger et al. (2011) suggest that diffuse pollution and runoff is likely to increase riverine contaminant loadings, but high discharges may dilute concentrations. Decreased travel time through alluvial systems is likely to result in less degradation of contaminants. Ray et al. (2002b) investigated the impact of very high flood flows on riverbank filtration sites using a combination of modelling and monitoring work. They concluded that combinations of pumping rate, riverbed hydraulic conductivity, contaminant properties and river stage are significant in controlling transport of contaminants to collector wells. Levy et al. (2011) investigated the impact of storm events on riverbed hydraulic conductivity and determined that storms have little impact on the overall filtration capacity. Mutiti and Levy (2010) showed that riverbed hydraulic conductivity is likely to increase during storm events due to the removal of fine sediment on the riverbed, but that the changes are small and do not pose a water quality risk. Wett et al. (2002) used riverbank monitoring and dynamic modelling to determine the hydraulic impact of flood induced infiltration on a riverbank filtration well. It was determined that during a period of high water levels, seepage to the collector well increased. After the event, seepage rates decreased due to increased groundwater recharge from both rainfall and stream infiltration and decreased river stage. In subsequent weeks well operation had depleted this storage and the seepage rate returned to steady state.

Understanding the resilience of existing riverbank filtration systems to climate change is critical to maintain security of public water supply in the future. Public water supply assets form part of society’s critical infrastructure (Water Services Regulation Authority (Ofwat) (2010); United States Environmental Protection Agency (2010)). As such, a working knowledge of the behaviour and performance of these assets during extreme events is of great importance to water managers, decision makers and the wider public (Simpson, 2014). Sharma and Amy (2009) and TECHNEAU (2009) identified that riverbank filtration systems are underutilised in developing countries and could be an effective sustainable water treatment technology in the future. An understanding of the potential impacts of climate change on prospective future RBF sites in these settings is critical for cost-effective investments in water infrastructure assets. Whilst numerous studies have detailed the impacts of storm events and high river flows on RBF systems, little work has been undertaken to understand the impact of full floodplain inundation of RBF systems from extreme flood events (Farnsworth and Hering, 2011). The objective of this paper is to characterise the water quality impacts of inundation of riverbank filtration systems by extreme flooding and the controls on recovery in water quality following such an event.

2. Materials and Methods

2.1. Study Site

The site is located by the River Thames in West London, England (Figure 1). The site was chosen on the basis of the following criteria: (1) easy and rapid access to the wells during and after a flooding event, (2) regular observations of floodwater levels and water quality during a flooding event (Addison, pers. comm.) and (3) continuous abstraction data during the flooding event. River flows are predominantly derived from groundwater discharge (baseflow index = 0.66, (National River Flow Archive (2014))) from the carbonate Chalk and Limestone aquifers located upstream. The principal aquifer at the RBF site is the Shepperton Gravels which have high transmissivity and storage ($T \approx 1400\ m^2/day$, $S \approx 0.2$ (dimensionless) (Naylor (1974), Vivendi Water Partnership (2002)). Borehole logs indicate the gravels have an average thickness of 5 m on the site. The gravels are overlain by approximately 1 m of well drained calcareous topsoil with a low organic carbon content (Cranfield University, 2015). Patchy clayey sands of relatively low permeability are also present. This physical and chemical soil composition indicates that any changes in the hydrochemistry of floodwater occurring during infiltration are likely to be small. The gravels are underlain by low permeability

London Clay. Recharge to the Shepperton Gravels is derived from both conventional rainfall-recharge mechanisms and riverbank infiltration induced by groundwater abstraction.

The site consists of a horizontal collector well system which is perpendicular to the river Thames. Three pump shafts are connected to a horizontal adit. Abstraction from the collector wells depresses groundwater levels and induces flow from River Thames and the gravel aquifer, as shown from estimated groundwater flowpaths (Vivendi Water Partnership, 2002) in Figure 1 (c). The nature of the pump shaft system results in a baseline water quality which varies along the adit. At Well 3, closest to the river, a river water signature is present which is affected by hyporheic zone processes. At Well 1, furthest from the river, a more groundwater dominant water quality is present. The site is licensed to abstract up to 40.91 Ml/d from the gravel collector wells. The collector well pumps are variable speed drive and have been protected to a flood design criteria of a 1 in 100 year flood event with 20% freeboard to account for climate change. There is an associated river abstraction and treatment works and all water undergoes extensive treatment.

Under normal operational conditions, groundwater is pumped directly into a membrane filtration plant then blended with partially treated surface water, before passing through a granular activated carbon (GAC) plant and subsequent disinfection and into supply. In times of inundation, the raw groundwater can be directed to a small reservoir, where it then follows the full surface water treatment process, avoiding the membrane filtration process and resulting in no impact on treated water supply.

2.2. Flooding Event and Monitoring Network

The flooding event used to determine the impacts of inundation on riverbank filtrate water quality occurred during January to February 2014. Winter rainfall for Southern England was 20% greater than the previous maximum in 1914/15 and the highest winter runoff total was recorded in the

Thames since records began in 1883 (CEH, 2014). Actual flows in the Lower Thames at Kingston were the highest since 1974 at 524 m^3/s. Flows have exceeded this rate 8 times over the record since 1883, which corresponds to an approximate return period of 1 in 17 years. Whilst this return period is not particularly high, the flooding was exceptional in duration (Huntingford et al., 2014). Flows at Kingston continuously exceeded 250 m^3/s for 76 days, over twice the previous longest period of 30 days in 1947 (Huntingford et al., 2014). Substantial surface inundation along the Thames was observed from Datchet, Berkshire to Shepperton, West London and was widely reported in the international media. At the study site, inundation was estimated to occur for 7 days based on daily site walkover visits by the site hydrological engineer (Addison, pers. comm.) and 15-minute river level data.

In order to determine the impacts of inundation on water quality, a groundwater and surface water monitoring network was set up (Figure 1). Table 1 details the available points. Daily rainfall data from Shepperton Lock, 3.3 km south east of the study site was used (Met Office, 2014). Daily river flows were recorded by the Environment Agency 5 km upstream of the site at Staines. Existing telemetry was used to record changes in abstraction rate, turbidity and groundwater level every 15 minutes through the inundation event at the collector wells. River level and water quality determinants (turbidity, dissolved organic carbon, specific electrical conductivity) were also recorded at the same frequency. Pumped spot water quality samples were taken at 8 intervals after the inundation at the collector wells and from the river. Samples were taken initially at a weekly interval for 5 weeks and then decreased to fortnightly and subsequently monthly with the last sample taken in June 2014. This allowed for the majority of the recovery in water levels and quality to be monitored. Historic water quality data from 2012 onwards was used for comparison with the event data.

2.3. Water quality sampling and analysis

Samples were taken from sample taps for each of the 3 wells and directly from the river. Additional sampling was also undertaken throughout the monitoring period at a combined sample point. This sample point is located immediately prior to the membrane filtration plant and is used to assess the water quality of the mixture of 3 wells before treatment. This point is an integrated flow-weighted sample of wells 1, 2 and 3. Prior to sampling, water samples were passed through a flow cell until hydrochemical parameters (temperature, dissolved oxygen, specific electrical conductivity) stabilised. Samples for dissolved organic carbon, fluorescence and absorbance analysis were filtered using 0.45 µm silver filters into acid washed glass vials. Analysis was undertaken using the methods detailed by Lapworth et al. (2009). Samples for inorganic analysis were filtered using 0.45 µm cellulose nitrate filters into Nalgene bottles. Chlorofluorocarbon (CFC) samples were collected and analysed using the methods reported in Gooddy et al. (2006). Samples for emerging organic contaminants were collected unfiltered into 1 litre glass bottles. Emerging organic contaminant analysis was undertaken by the UK Environment Agency National Laboratory Service with a multi-residue gas chromatography-mass spectrometry (GC-MS) method screening for over 1000 organic compounds as detailed by Sorensen et al. (2015). This method gives detection limits of 0.01 to 0.1 µg/L for 90% of compounds and a reporting limit of 0.01 µg/L for 75% of compounds. Microbial samples were collected unfiltered and analysed using a pour-plate method. All samples were kept in darkness at 4 ^{o}C prior to analysis. All fluorescence data was corrected for inner filter effects using the corrected absorbance data (Lakowicz, 1983). The data were reported in standard Raman units, which normalises the intensity by the area under the Raman peak between emission wavelengths 380-410 for the excitation wavelength of 348 nm. Post processing of fluorescence data was carried out using an R script described by Lapworth and Kinniburgh (2009) within the statistical package R.

2.4. Estimation of collector well water sources

The relative significance of different sources of water to the collector well system through the flood event was quantified using both hydrochemical and physical approaches. Binary mixing models were used to derive estimates of the fraction of surface water (F_{sw}) for the gravel wells. The river concentration data was used as one end-member and baseline concentrations (as estimated in June 2014) at Well 1 were used to represent the groundwater end-member.

Estimates of F_{sw} were compared against a simple spreadsheet model developed to estimate the proportion of total abstraction derived from inundation water, conventional riverbank filtration and conventional recharge/gravel storage depletion on a daily timestep. Flow to the gravels from the river by conventional riverbank filtration (Q_{RBF}, m^3/day) is estimated using a Darcy flux based on the observed head gradient ($h_r - h_{aq}/x$, unitless) between the river and gravel observation boreholes, a cross sectional area of flow (A_{RB}, m^2) and an estimate for riverbed permeability ($K_{RB,}$ m/day):

$$(1)\ \ Q_{RBF} = K_{RB}.A_{RB}.\frac{h_r - h_{aq}}{x}$$

Riverbed permeability estimates were derived from previous groundwater model calibration for the site by Vivendi Water Partnership (2002) and from local grain size analysis by Naylor (1974). The head gradient was estimated based on daily observed groundwater and river levels at the study site. Flow to the gravels by inundation (Q_{IND}, m^3/day) is estimated using a simple water balance approach considering the timing and amount of inundation at the site:

$$(2)\ \ Q_{IND} = \frac{dh_i}{dt}.A_{IND}.f_{IND}$$

Where dh_i/dt (m/day) is the change in inundation water level through time, A_{IND} (m^2) is the estimated area of inundation contributing to flow to the wells and f_{IND} is a calibration factor which allows for inundation water to be lost by other means such as evaporation and flow back to the river. Table 2 details the values used Equations 1 and 2. The change in inundation water level is derived from a linear decrease in water level based on daily site observations which indicated that the maximum water depth on site was 0.6 m and this took 7 days to recede (Addison, pers. comm.). A_{RB} and A_{IND} were estimated based on previous groundwater model collector well capture zones (Vivendi Water Partnership, 2002) and the estimated area of inundation (0.2 km^2, Addison (pers. comm.)).

Under normal conditions, river levels at the study site are heavily controlled by the environmental regulator through upstream level management structures to allow navigation. Consequently, normal variations in river flow do not result in significant differences in river water level, water depth and channel cross-sectional area (Hinks, 2013). Consequently, for the purposes of calculating the flow to the gravels from conventional riverbank filtration under normal conditions (i.e. not from a flood), it was assumed that the cross sectional area of the river was constant through time. Direct quantitative measurements of floodwater flows back to the river and evaporation during an extreme flood event is highly challenging and dangerous. Consequently, f_{IND} was initially estimated with a heuristic approach using expert hydrogeological judgement based on the site hydrogeology and daily site observations that suggest that half of the inundated water evaporated or flowed back to the river (Addison, pers. comm.). There is likely to be considerable uncertainty in the parameterisation of f_{IND} and consequently for the purposes of spreadsheet modelling a range of 0.3 – 0.7 was used. Increasing the value of f_{IND} results in more of the abstracted water being drawn from floodwater relative to bank filtration and gravel storage. It should be noted that for modelling purposes, the approach adopted to estimate Q_{IND} assumes that water that is infiltrating immediately contributes to groundwater flow to the gravel well. In reality it is likely there is some delay between any vertical infiltration through the clayey sands and topsoil to the saturated zone and to the abstraction from the gravel wells and consequently the additional water contribution from inundation is likely to be dispersed through time. The impact of this model limitation is discussed in section 3.2.5. The total flow to the gravel wells, Q_t (m^3/day), can be estimated as:

$$(3)\quad Q_t = Q_{RBF} + Q_{IND} + Q_{GWR}$$

Where Q_{GWR} (m^3/day) is the additional flow to the gravel wells which is from conventional recharge and groundwater storage. As Q_t was known a priori from recorded abstraction data, Q_{GWR} was back-calculated during the modelling process.

3. Results and Discussion

3.1. Hydrological Context, Impacts of Flooding and Recovery

Figure 1 (c) shows the best approximation of the spatial extent of inundation of the site based on the site walkover visits (Addison, pers. comm.) which has been estimated as 0.2 km^2. Figure 2 shows the context of the flooding event in relation to the previous year's hydrology and hydrochemistry. The 2012 – 2014 period was hydrologically exceptional (Marsh et al., 2013). The 2010-12 drought ended with a transition to flood. Following increases in river flows during winter 2012/13 and a return to long term average conditions through much of 2013, flows began to increase rapidly to above long term average values in December 2013.

Figure 3 presents the hydrometric data collected before, during and after the flooding event. Substantial rainfall of up to 30 mm per day occurred between December 2013 and February 2014. This resulted in large amounts of runoff in the Thames catchment resulting in increases in river flows up to 320 % of long term average (LTA) values in February 2014. Following this peak, river flows decreased back to long term average values by April 2014. Large rises were also observed in river stage and groundwater levels in the gravels. As shown in Figure 3 (c), both pumping groundwater levels in the collector system and abstraction-impacted observation borehole levels remained below the river level throughout the period, even during the inundation event. This results in a continuous head gradient and corresponding flux of water from the river to the gravel well system both laterally through a RBF mechanism and vertically during the inundation event.

Observation borehole data indicate that during the peaks in river flows in January and February 2014, groundwater levels at the site were below the ground surface, therefore the gravel aquifer and the inundation ponded water were hydraulically disconnected. Consequently infiltration of this water into the groundwater system occurred through gravity drainage and independent of groundwater abstraction. However, during the inundation ponded water did not directly enter the collector wells via the pump shafts. During the flooding event, total abstraction from the gravel wells was increased from a base load of approximately 20 Ml/day to a peak of 40 Ml/day. This increase in abstraction was primarily the result of the combined operation of all 3 wells at approximately 13 Ml/day each. After the event, abstraction at Well 2 was intermittently reduced.

3.2. Hydrochemical Impacts of Flooding and Recovery

3.2.1. Hydrochemical Context

Figure 2 shows the hydrochemical context of the flood event. Dissolved Organic Carbon (DOC) and Specific Electrical Conductivity (SEC) data for the combined sample point indicate the hydrochemical impact of this extreme event. DOC increased to approximately 3.5 mg/l on 19th February 2014, in comparison to long term average (LTA) values of 2.64 mg/l. The 2014 flood event corresponds to an increase of 132% relative to long term average values. SEC decreased to approximately 517 µS/cm on 19th February 2014, in comparison to long term average values of 646 µS/cm. The 2014 flood event corresponds to a decrease of 80% of long term average values. These trends are associated with a greater fraction of high DOC and low SEC concentration surface runoff in both the Thames and riverbank filtrate, relative to more mineralised groundwater inputs. This dilution of groundwater inputs by surface runoff and resulting high river flows, corroborates with the scenarios developed by Sprenger et al. (2011). After the flood event, DOC and SEC data from the combined sample point recover to 102% and 96% of long term average values respectively. Data from the individual wells also stabilise following the flood event. This suggests that the sampling effectively captured the majority of the recovery in water quality back to more normal conditions. It should be noted whilst concentrations stabilise after the flooding event, there is still some uncertainty in the recovery back to baseline conditions by the final sampling campaign in June 2014 for other parameters where pre-event concentrations are not known. Baseline data for the three wells differ from the combined sample point data, which is a result of different sampling and analytical methodologies for these data sets. River flows also returned to long term average values.

3.2.2. Rapid response determinands – turbidity and microbial detects

Figure 4 shows turbidity data taken from 15-minute telemetry for the River intake and the wells and microbiological spot samples from the combined sample point for the 3 wells. River turbidity shows a moderate correlation with river flow (R^2 = 0.50 for daily data for period 1st January 2014 – 1st June 2014, see supplementary information Figure S1) as runoff events contribute particulate loadings to flows. The impact of inundation events on the gravel wells can be observed in the turbidity data. In January 2014, high turbidity (>50 nephelometric turbidity units (NTU)) is observed in the river. However, site inundation did not occur and turbidity in the gravel wells remained relatively low (<0.5 NTU). In contrast, during February, rapid spikes in turbidity (up to 1.5 NTU) occur in the gravel wells, which is an order of magnitude lower than river values (50 NTU). This rapid response indicates that there is a fast pathway for floodwater to reach the gravel wells, which is likely to be through vertical infiltration through the soils into the gravel aquifer. However, the substantial reduction in turbidity observed in comparison to river water, suggests that there is still significant attenuation occurring in the shallow topsoil and clayey sands. The increase in groundwater abstraction rates during the inundation event is likely to have increased the speed of recovery in water quality by pumping out any floodwater that has infiltrated under gravity and diluting it with gravel groundwater. Increases in microbial detects are also observed, with peaks of up to 4 colony-forming units (cfu)/100ml for

E.coli in the gravel wells. These values are up to 3 orders of magnitude smaller than the values for the river. It is suggested the observed increases are the result of a combination of vertical infiltration and conventional riverbank filtration, although this is uncertain due to data paucity.

3.2.3. DOC, Organic Contaminants and Dissolved Oxygen

Figure 5 shows DOC, total micro-organic detects, Specific Ultraviolet Absorbance (SUVA), and dissolved oxygen (DO) for the gravel wells and the river through the inundation event. Dissolved oxygen in the River Thames shows an increasing trend from 2.1 to 8.2 mg/l following the flooding event. This reflects a reduction in riverine DOC loading from 15 mg/l to 5 mg/l and consequently a reduction in microbial consumption of DO. Immediately after the inundation, DO concentrations in the RBF system wells were high at an average of 4.1 mg/L. The average baseline DO concentration in June 2014 was 0.93 mg/L. DO concentrations immediately after flooding correspond to 440% of baseline concentrations. This is likely to be the result of a combination of direct floodwater infiltration, rapid-rainfall recharge and flushing of the unsaturated zone as groundwater levels rise. Decreases in dissolved oxygen in the RBF system wells reflect a reducing influence of these processes at the site through time. Decreases occur relatively rapidly during the first few weeks following the flood event, with average well DO concentrations falling to 2.6 mg/L (280% of baseline concentration) and 1.8 mg/L (190% of baseline concentration) after 1 and 2 weeks respectively. These decreases are likely to be controlled by both the rate of lateral groundwater flow within the gravels and the increased abstraction rate. By abstracting at a higher rate, any floodwater and rapid rainfall-recharge that has infiltrated into the groundwater system can be pumped out and diluted with gravel groundwater and riverbank filtrated water. Increases in DO of 0.3 – 0.5 mg/l can be observed in wells 1 and 2 during the 4th sample round which coincides with a reduction in abstraction at well 2. It is postulated that this reduction in abstraction resulted in relatively less low-

DO concentration groundwater being drawn into the collector well from the gravels in comparison to the high-DO concentration water derived from recharge. Overall, Well 1 has the lowest DO for most of the recovery which is likely to be a reflection of background gravel groundwater quality. Well 3 shows the largest decrease in DO (from 4.2 to 0.4 mg/l) which is likely to reflect the transition from rainfall-recharge, floodwater infiltration and unsaturated zone flushing to drawing water from a less oxic hyporheic zone near the river through a conventional RBF mechanism.

DOC data for the gravel wells show mean concentrations decreasing from 3.1 mg/L immediately after inundation to 2.5 mg/L 5 weeks later. Baseline concentrations in June 2014 are estimated to be an average of 2.23 mg/L. These changes correspond to a decrease from 140 to ≈110% of baseline values over the first 5 weeks. Decreases are also observed in the river as flows return to normal average conditions. Changes in DOC in the gravels are likely to be the result of two factors: (1) decrease in DOC in the river which bounds the system, (2) floodwater infiltration during the inundation period (7 days). The highest DOC values are observed at Well 1 which is likely to reflect localised sources of organic carbon such as nearby landfills and Golf Courses. The ratio of indices of Tryptophan-like and Fulvic-like fluorescence of organic matter have been shown to be a useful tracer of sources of organic carbon in groundwater and surface water systems (Lapworth et al. (2008); Baker (2001)). Tryptophan:Fulvic ratio data at the study site suggest there is a different source of DOC at Well 1 and Well 2 than in the river during baseline conditions (1Figure S2). The large decreases in DOC at Well 3 (3.0 to 2.0 mg/l) are likely to reflect the transition from floodwater infiltration to water that has been subject to DOC degradation in the hyporheic zone through the normal RBF process. The Specific Ultraviolet Absorbance (SUVA) of organic carbon provides an indication of the aromaticity of the organic carbon (Weishaar et al., 2003) which can result in formation of disinfection byproducts (DBPs) (Singer, 1999). SUVA data indicate that during the first few weeks after the inundation event, the aromaticity of DOC in the river is high (SUVA = 3.5 L/mg-

M). This is likely to have a significant impact on formation of disinfection byproducts (DBPs) if the water was to be chlorinated without DOC removal. SUVA values for Well 1 – 3 in the first 2 sample rounds are relatively low at 2.42 – 2.79 L/mg-M.

Riverine emerging micro-organic detects increased from 5-7 detects to 15-17 detects following the inundation event. This increase in detections of up to 300% reflects reduced dilution as river flows decrease. The emerging organic contaminants detected are from a broad range of classes; pesticides, herbicides, personal care products and plasticisers. The insect repellant N,N-Diethyl-m-toluamide (DEET) and the herbicide propyzamide were detected 8 and 6 times, respectively, in the gravel wells at concentrations up to 0.02 µg/l. The anticonvulsant drug carbamazepine was detected 7 times in the wells at concentrations up to 0.04 µg/l, both DEET and carbamazepine been shown to be found frequently persist in groundwater (Lapworth et al., 2012). In the river, DEET was detected in every sample at concentrations up to 0.12 µg/l and Caffeine and Tetraacetylethylenediamine (TAED) were also regularly observed (6 and 7 detections and maximum concentrations of 0.18 and 0.17 µg/l respectively). These compounds have also been reported in groundwater in a number of studies and again reflect their persistence and use as tracers of surface water- groundwater mixing (Sorensen et al. (2015); Stuart et al. (2014); Engelhardt et al. (2011); Buerge et al. (2003)). In general, detects in the gravel wells decrease through time, reflecting a decrease in influence of flood water infiltration. Towards the end of the monitoring when baseline conditions had resumed, total organic detects in the river are over 3 times greater than those observed in the gravel wells. This implies that under conventional operating regimes and river levels at long term average (LTA) values, the colmation layer in the hyporheic zone and the flow path through the aquifer to the gravel wells are able to attenuate some of these types of compounds. This is likely to be the result of a number of processes such as mixing and sorption in the aquifer and sorption and biological degradation in the colmation layer (Stuart et al. (2014); Lewandowski et al. (2011)).

3.2.4. SEC, Nitrate,CFC-11 and CFC-12

Figure 6 shows specific electrical conductivity (SEC), nitrate and trichlorofluoromethane (CFC-11) concentrations for the gravel wells and the river. A general increase in SEC is observed through time in the river from 400 to 600 µS/cm. This reflects a return to a more baseflow-dominated flow regime with higher fractions of mineralised groundwater inputs from the Chalk and Limestone aquifers relative to runoff. These increases are also observed in the wells, with an average increase from 620 to 660 µS/cm over the first 3 sample rounds relative to an average baseline SEC of 686 µS/cm. This increase from 90 to 96% of the baseline SEC reflects two processes: (1) increased mineralisation of the riverbank filtrate due to a higher baseflow component in the river, (2) increased abstraction of gravel groundwater and riverbank filtrate relative to any low mineralisation floodwater infiltrate. Well 1 and 2 show consistently higher mineralisation (baseline SEC = 700 µS/cm) relative to Well 3 (baseline SEC = 650 µS/cm), which reflects both the impact of gravel groundwater on the wells further away from the river and mixing with bank-filtrated river water at Well 3.

Impacts of the abstraction regime in the gravel wells can also be observed. During the 4th and 5th sampling round, as abstraction at Well 2 was reduced, a decrease in SEC of 50 µS/cm can be observed at this well (Fig. 6b). It is likely that during this period, Well 2 is no longer drawing mineralised groundwater from the aquifer, but is just pumping residual water associated with the recharge and floodwater infiltration from within the collector well system, resulting in a decrease in SEC. During the 6th to 8th sampling rounds, SEC appears to increase again without any increase in abstraction. It is likely that by this time, the RBF system has returned to a hydrochemical quasi-steady state with limited residual influence of direct floodwater infiltration.

Nitrate trends reflect the influence of the river on the RBF system, with higher concentrations in the river and at Well 3 than at Wells 2 and 1 (Fig. 6c). Nitrate in the river and at Well 3 increases to stable concentrations of 25 mg/l and 20 mg/l, respectively, in approximately 6 weeks. This is associated with an increased proportion of nitrate-rich baseflow within the Thames from upstream discharge from chalk and limestone aquifers. Despite the decrease in DO through time observed at Well 3, no substantial decreases in nitrate are observed associated with denitrification. It is likely this is the result of two factors: (1) the low concentration of organic carbon substrate as evidenced by the low DOC values (≈2.2 mg/l), (2) a limited microbial community for denitrification as result of the flooding. Well 2 and Well 1 generally show stable trends between 5 and 10 mg/l which reflect low background nitrate concentrations in the gravel groundwater.

CFC-11 and dichlorodifluoromethane (CFC-12) concentration data show broadly similar temporal and spatial trends which indicates that preferential CFC degradation is unlikely to be occurring (Figure S3, R_2 = 0.64). All CFC data give modern fraction values > 1. This “over-modern” data cannot be used as groundwater dating tool, however they can be used as tracers to understand mixing processes. Concentrations of CFC-11 (Figure 6d) show the extent of river water influence on the RBF system. Riverine CFC-11 concentrations fall rapidly initially which is likely to reflect a transition from river flows controlled by flood runoff to one dominated by relatively unpolluted groundwater from the chalk and limestones. There is likely to be a lag between recharge of flood water to these upstream aquifers and subsequent discharge of this polluted water to the river. It is plausible this lag is the cause of the second observed increase in CFC-11 concentrations, with discharge of shallow polluted groundwater in the chalk and limestones to the river. As this polluted groundwater discharges out of these aquifers, CFC-11 concentrations fall again. This trend observed in the river is clearly visible in Well 3 but is attenuated in Wells 1 and 2.

The use of CFC data to derive estimates of groundwater ages is well established and over-modern CFC concentration data have been used for groundwater tracing (Darling et al., 2012; Darling et al., 2010). However, there has been limited application of this data to surface waters. This novel application of CFC-11 concentration data to estimate sources of water to the river has potential to be a useful tool for future water resource management.

3.2.5. Estimation of collector well water sources through flooding

Figure 7 (a) shows estimates of the breakdown of total abstraction Q_t from riverbank filtration Q_{RBF}, inundation Q_{IND} and conventional recharge and gravel storage Q_{GWR}. The model indicates that the proportion of riverbank filtrate to the collector well system is approximately 40 to 70% of the total abstraction. The relative increase and subsequent decrease in the contribution of riverbank filtrate is primarily controlled by the change in the hydraulic gradient between the gravel wells and the river. It can be observed that during the inundation period, modelling suggests that between 15 and 44% of the total abstraction can be derived from the infiltrating flood water for f_{IND} = 0.3 – 0.7. Increasing f_{IND} by 0.1 increases the relative contribution of floodwater to total abstraction by 5.2 – 6.2%. As discussed in section 2.4, it is highly likely that this input of water is temporally dispersed rather than instantaneously entering the collector well system due to lag in infiltration through any clayey sands. Consequently, this percentage contribution is likely to be lower in reality but may persist for longer. As there is an unsaturated zone present above the water table at the site (Figure 2), flood water infiltrated under gravity drainage. As the collector well system and the flood waters are hydraulically disconnected, increasing abstraction during and after the inundation period will draw more gravel groundwater into the wells and dilute any surface infiltration. The flexible operating regime at the site resulted in increased abstraction during the inundation event. This is likely to have mitigated the hydrochemical impact of the inundation to some degree through increasing dilution by gravel groundwater.

Figure 7 (b) shows estimates of the average fraction of surface water for the collector wells as derived by nitrate and CFC-11 data. Data for these determinands for Well 1 and the river reflected distinct end-members for the collector well system. Chloride data was not used as Well 1 and the river did not suitably reflect end-members of the system. A poor correlation with sodium data was observed (R^2 = 0.06). This implies that multiple sources of chloride and sodium were present which limits the use of simple binary mixing models. The fraction of surface water at Well 3 (F_{sw} = 0.5 – 0.75) corroborates well with estimates of riverbank filtrate contributions to flow derived from modelling previously discussed. The fraction of surface water at Well 2 or 1 (F_{sw} = 0 – 0.3) is significantly lower reflecting a greater contribution of gravel groundwater. At Well 1 and 2 decreases in F_{sw} are observed from 0.2 – 0.3 during the first two sampling rounds to around 0 - 0.1 during the last two samples. These decreases are relatively small and are likely to reflect the limited residual influence of any floodwater infiltration and direct recharge. The relatively stable mixing ratios in the final two sampling rounds are likely to represent the proportions of water in the collector well system derived from RBF and gravel groundwater under normal conditions. Further research comparing the two methods used here with other hydrological and mixing models would also be beneficial, but is considered to be out of scope of the current study.

3.3. Conceptual model of flood recovery

Figure 8 gives a conceptual model of the impact and recovery from flooding observed at the site. The impact of the inundation event on the gravel groundwater wells can be characterised by the following: (1) high DOC, turbidity, DO, micro-organic and microbial contaminants associated with floodwater infiltration, recharge and unsaturated zone flushing, (2) Low SEC due to reduced groundwater component, (3) Increased fraction of surface water (F_{sw}). The recovery from flooding is characterised by transition to a regime dominated by two end-members, a landside groundwater component at Well 1 and riverbank-filtrated component at Well 3 with: (1) Increased SEC (2) Decreased DOC, DO, turbidity, micro-organic pollutant detects, (3) Rapid decreases in microbial detects and turbidity, (4) Lower F_{sw}. The speed of the recovery is constrained by the site's hydrogeological setting, the abstraction regime and the background water quality trends at site boundary conditions. The relatively low permeability of the clayey sands overlying the gravel aquifer is likely to attenuate direct floodwater inundation to some extent. The high transmissivity Shepperton gravels allow any recharge and floodwater infiltration that does occur to move rapidly through the groundwater system. Additionally, the increased abstraction rates assist in diluting any floodwater that has infiltrated into the groundwater system. This is likely to have affected the recovery in terms of turbidity and microbiology. Whilst these processes may enhance the rate of recovery of the other determinands, the background trends observed in the river will be a significant control. Most floodwater impacts are observed within the first 2 – 3 weeks, with a return to baseline conditions within 6 weeks. Reductions in abstraction rates following the inundation, appears to slow recovery temporarily, as evidenced by the DO and SEC data.

This conceptual model is the first published assessment of the hydrological and hydrochemical impacts of extreme flooding at an RBF site and the subsequent recovery. Overall, the conceptual model is likely to be generic and broadly applicable to other sites. However, it is important to note that all RBF sites and associated catchments will have different site configurations, hydrological and hydrogeological properties. Moreover, all flood events will be different, with variations in antecedent conditions, rainfall intensities and distributions. Consequently, the hydrochemical impact and recovery from flooding will always vary to some degree for different flood events and different RBF sites. Further research building on this conceptual model through development of relationships between different flood events, RBF site configurations and the subsequent hydrochemical impact and recovery would be beneficial for management of RBF sites.

3.4. Implications for management and operation of other RBF Systems

This study has shown the importance of operational flexibility for RBF sites with limited aquifer thickness (<10 m) in mitigating the impacts of extreme floodplain inundation water quality, particularly with regard to turbidity and microbiology. By continuing to operate the site and increasing abstraction rates after flooding, rapid reductions in contaminant loadings have been achieved through increased dilution of surface infiltrate with gravel groundwater. This was possible at this location due to the configuration of the site infrastructure. If extreme flooding was to occur at a site without the operational resilience and flexibility of this study site, it is plausible that contaminant loadings associated with floodwater infiltration would be observed for longer periods of time. This has the potential to induce significant additional costs associated with: (1) treatment of the water from the wells and (2) increased abstraction elsewhere for blending if treatment options were not sufficient. These results have important implications for RBF system management in view of more frequent extreme events under climate change (Prudhomme et al. (2003); Fowler et al. (2005); Simpson (2014)). It is recommended that water managers adopt flexible operating regimes such as the one implemented at this study site, to increase resilience of shallow aquifer RBF systems under potentially more extreme climate scenarios. Such measures would include: (1) Regulatory flexibility to allow increases in pumping, (2) Variable speed drive pumps, (3) Flood-proofed infrastructure, (4) Sufficient treatment, network and storage capacity to handle increased volumes of water, (5) Suitable treatment processes to cope with different water qualities.

The study has also important implications for decision-makers considering the development of future RBF systems, particularly in developing countries. The role of the river water quality in the longer term recovery in the gravel wells for some parameters (DOC, SEC), highlights the importance of suitable monitoring and characterisation of hydrochemical boundaries to RBF systems. Whilst abstraction rates have affected the recovery from flooding, the high transmissivity of the Shepperton gravels has also facilitated a rapid recovery by allowing rapid transfer of infiltrating floodwater through the groundwater system to the abstraction wells. High transmissivities are also beneficial under drought conditions where collector well yields may be constrained by borehole water levels under pumping conditions. In these situations, higher transmissivities and consequently smaller drawdowns may provide significant additional water when borehole yields are constrained by low groundwater levels. However, under periods of normal operation, a more moderate transmissivity aquifer material may be more beneficial as increased travel times between the river and the wells allow for more contaminant attenuation. This highlights a difficult decision for water managers to consider and one which is the subject of recent research (UKWIR, 2014); whether to plan for the mean or the extreme? Under extreme conditions siting a RBF system in a high transmissivity formation may be most beneficial, but under average conditions a moderate transmissivity formation may be most effective for contaminant removal. This decision will ultimately be site-specific depending on the purpose of the site and will form part of a wider optimisation exercise considering technical, economic, regulatory and land use factors (Grischek et al., 2003).

4. Conclusions

This study has characterised the hydrochemical impact and recovery from extreme floodplain inundation at a RBF site of limited aquifer thickness. The controls on the recovery from the flooding have been explored and suggestions have been made regarding future design and operation of RBF systems in these settings. It is concluded that:

- RBF inundation in shallow aquifer settings is characterised by high turbidity, organic contaminant, microbial detects, DO and DOC, and low SEC. A rapid recovery is observed in turbidity and microbial detects and recoveries in other determinands take approximately 6 weeks.
- Recovery rates are constrained by a number of parameters. Rapid recoveries in turbidity and microbial detects are controlled by increased abstraction diluting floodwater that has infiltrated into the groundwater system. The high permeability of the gravels allows for rapid recharge and saturated transport of contaminants to the wells. Whilst increased abstraction is likely to have some impact, the long term changes in the hydrochemical boundaries to the system such as the river, are likely to be significant in controlling the water quality trends at the gravel wells.
- Whilst this conceptual model is broadly generic, different flood events and RBF site configurations will result in different hydrochemical impacts. Further research exploring these controls on flooding impacts will improve RBF site management.
- In order to mitigate against the hydrochemical impacts of floodplain inundation, it is recommended that RBF sites in shallow aquifer settings are operated flexibly with the capacity to vary abstraction when needed.
- For future prospective RBF sites, this study highlights the importance of developing a good conceptual understanding of hydrochemical boundaries and site hydrogeology. Such an understanding can only be developed through monitoring of the site under both baseline and flood conditions. Whether a site is hydrogeologically suitable will depend on the purpose of the site and will be part of a wider optimisation task.

Acknowledgements

The authors would like to thank staff at Affinity Water for facilitating this work and providing data, D. Allen and P. Williams (BGS) for assistance in hydrochemical sampling and analysis. This work was funded by BGS (NERC) National Capability Programme. The BGS authors publish with permission of the Executive Director, British Geological Survey (NERC).

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Table 2: Parameters used to estimate sources of water to the collector wells

Table 1
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Monitoring Location	Sample Data	Telemetry Data
River Intake	Total and dissolved organic carbon, specific electrical conductivity, E. Coli, Coliforms, Enterococcus, dissolved oxygen, total emerging organic detects, fluorescence/absorbance properties, major ions and inorganics, CFCs	River level, turbidity, dissolved organic carbon, specific electrical conductivity
Well 1 Well 2 Well 3	Specific electrical conductivity, dissolved organic carbon, dissolved oxygen, total emerging organic detects, fluorescence/absorbance properties, major ions and inorganics, CFCs	Abstraction Rate, Turbidity
Well 1, 2 and 3 Combined	Total organic carbon, specific electrical conductivity, *E. Coli* , Coliforms, Enterococcus	
Shepparton		Rainfall
Staines		River Flows and Levels

Table 2

Click here to download Table: Table 2.xlsx

Parameter	Value	Units	Reasoning
K_{RB}	30	m/d	Previous estimates of riverbed permeability used in groundwater modelling (VWP, 2002; Naylor 1974)
A_{RB}	27500	m^2	Length of collector well capture zone along the river (500 m, from VWP, 2002) and the width of the river (55 m)
$h_r – h_{aq}/x$	Varies daily based on observed water levels. Range 0.014 - 0.044	-	Observed groundwater and river levels at the study site
A_{IND}	2	km^2	Observation of inundation extent and collector well capture zone (VWP, 2002)
f_{IND}	0.3 - 0.7		Observations indicating up to half of inundation water may be lost to evaporation and back-flow to the river
dh_i/dt	0.09	m/day	Observations indicating maximum inundation of 0.6 m and 7 days for water levels to recede

Figure 1

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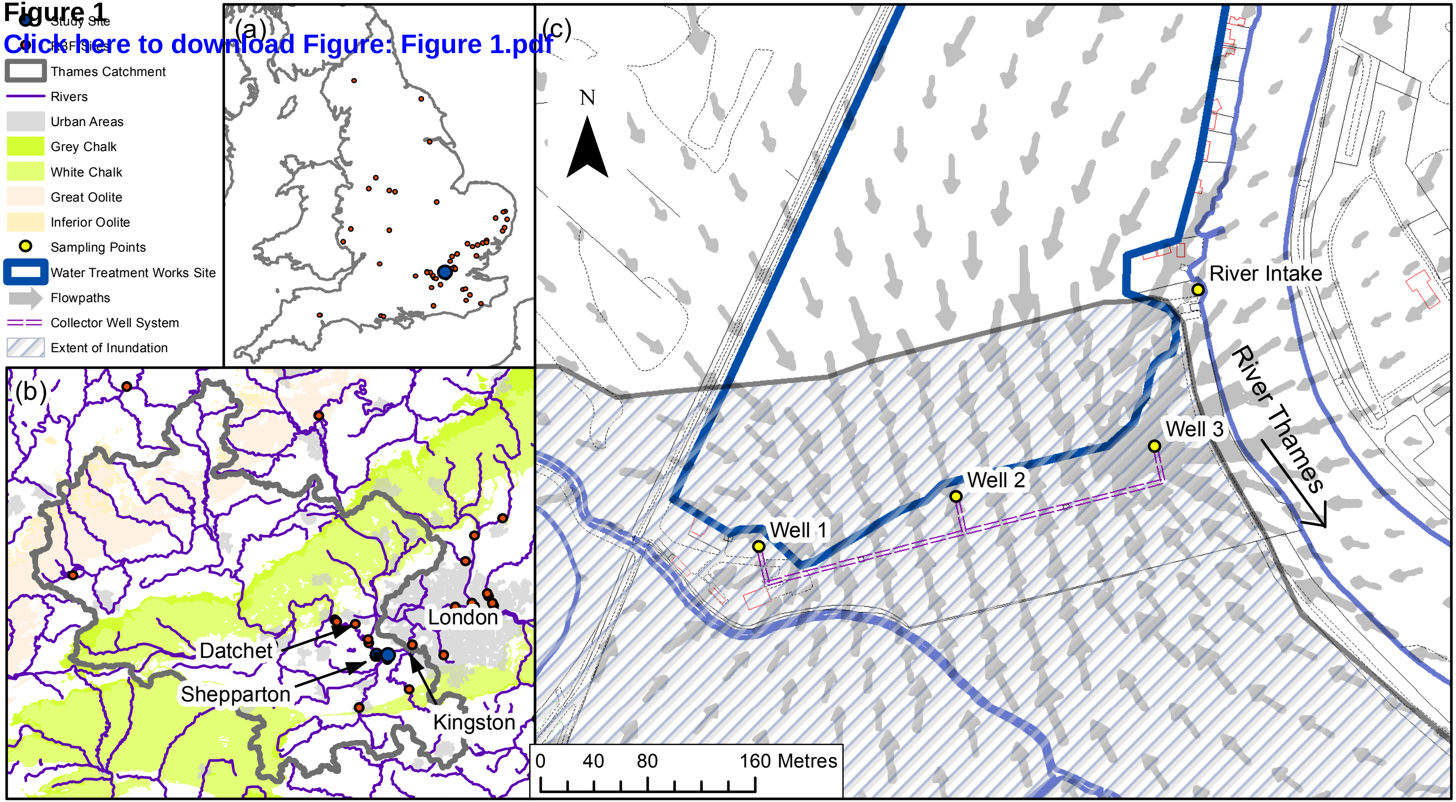

Figure 2

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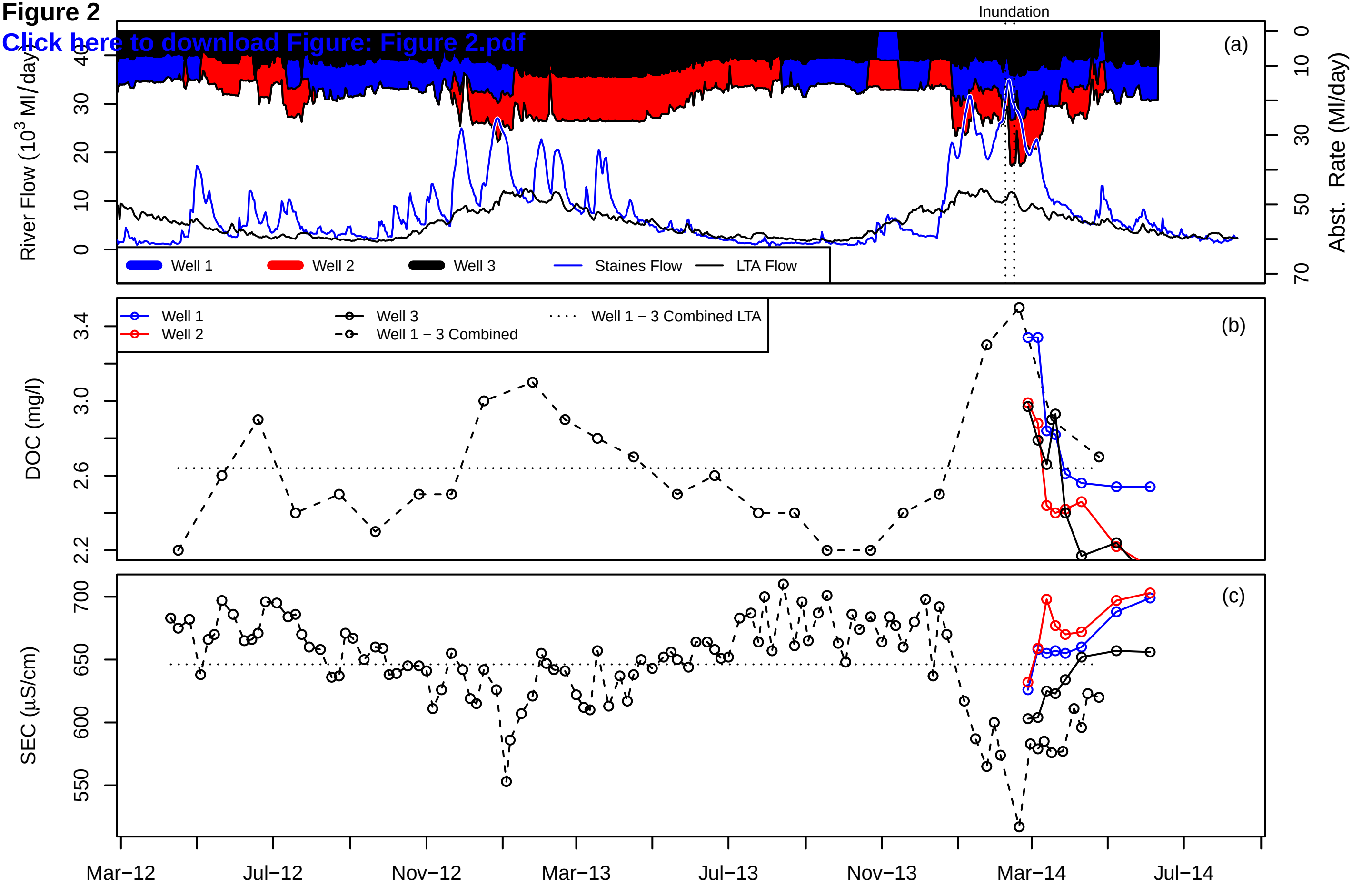

Figure 3

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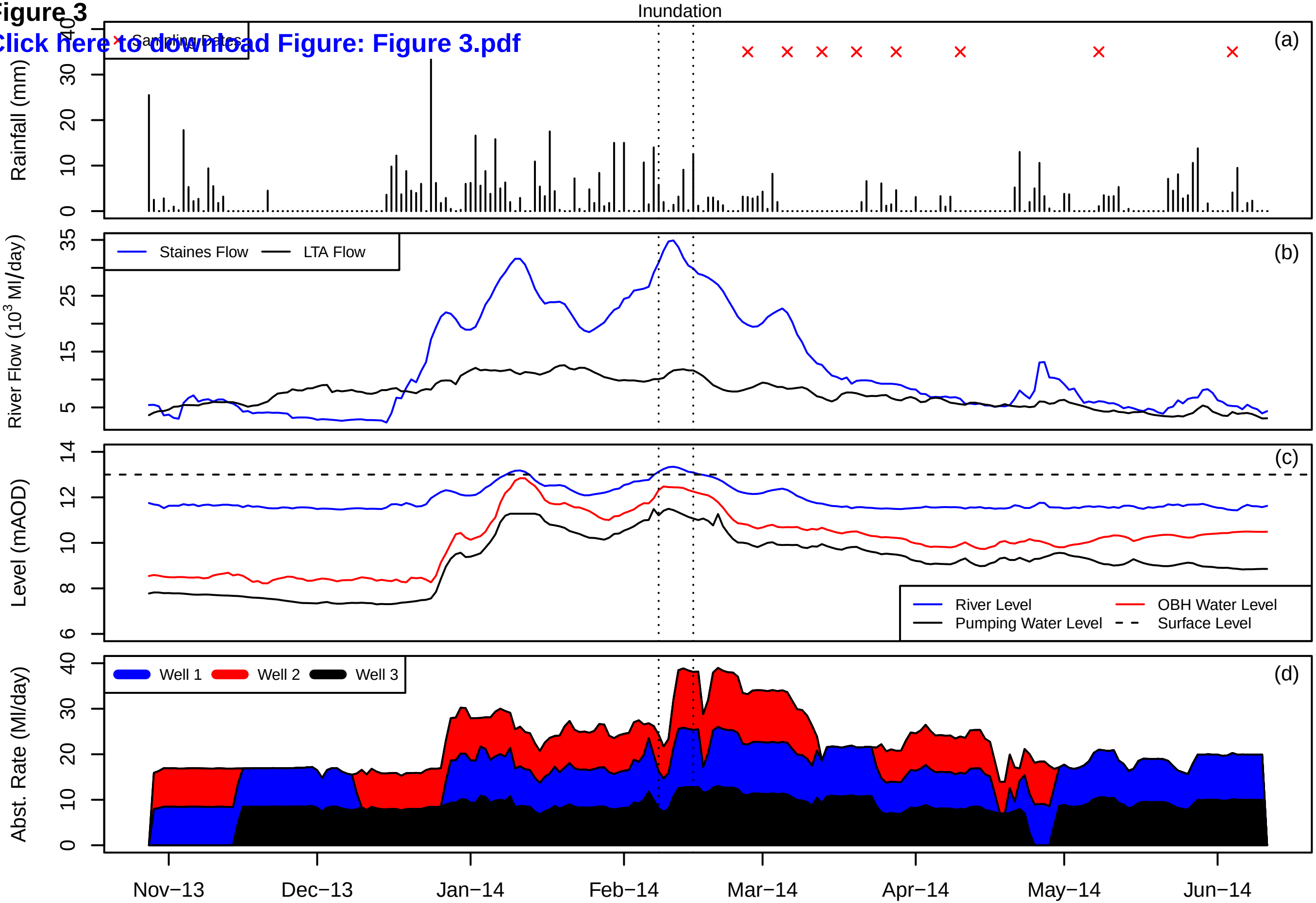

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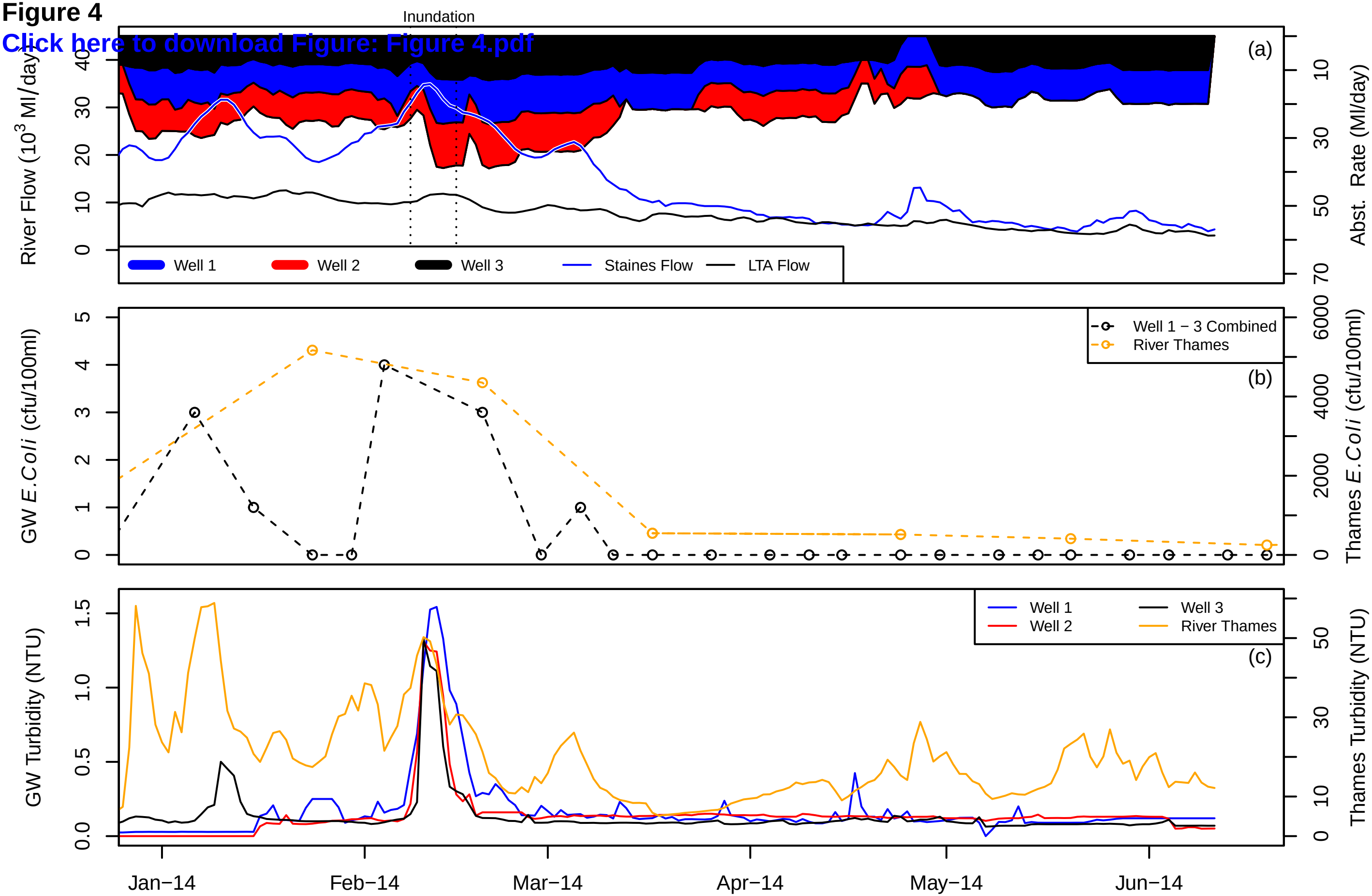

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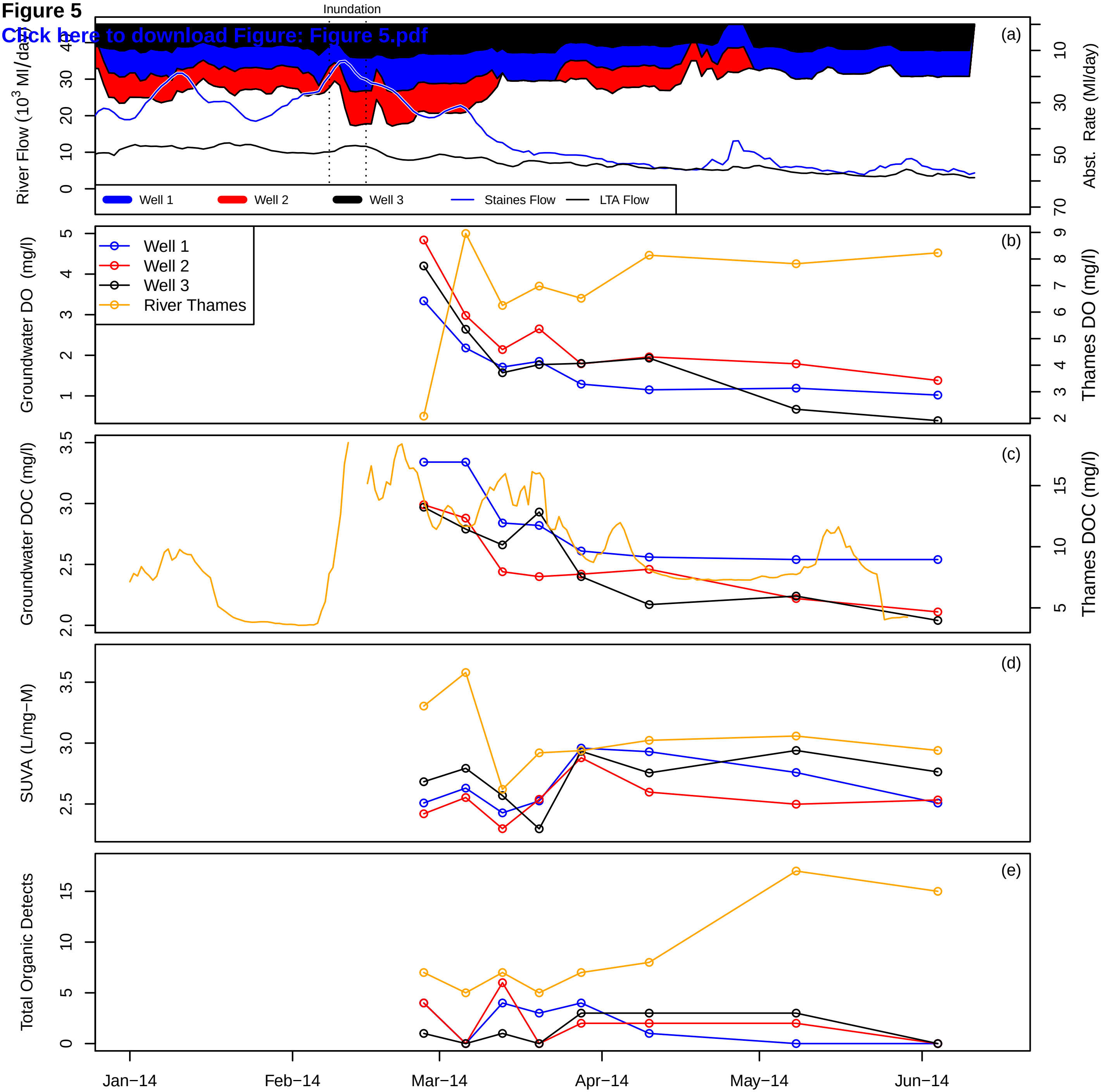

Figure 6

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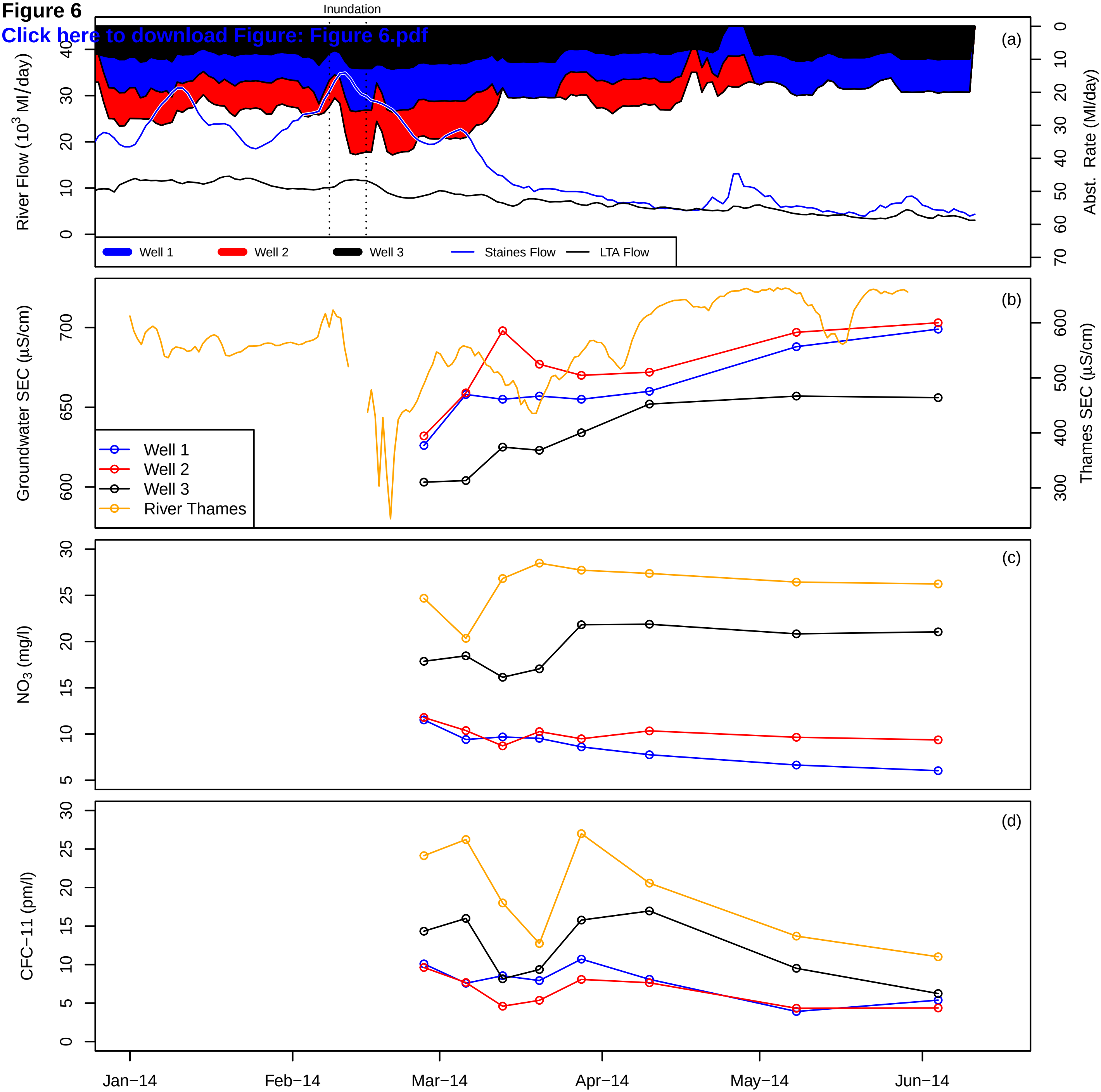

Figure 7
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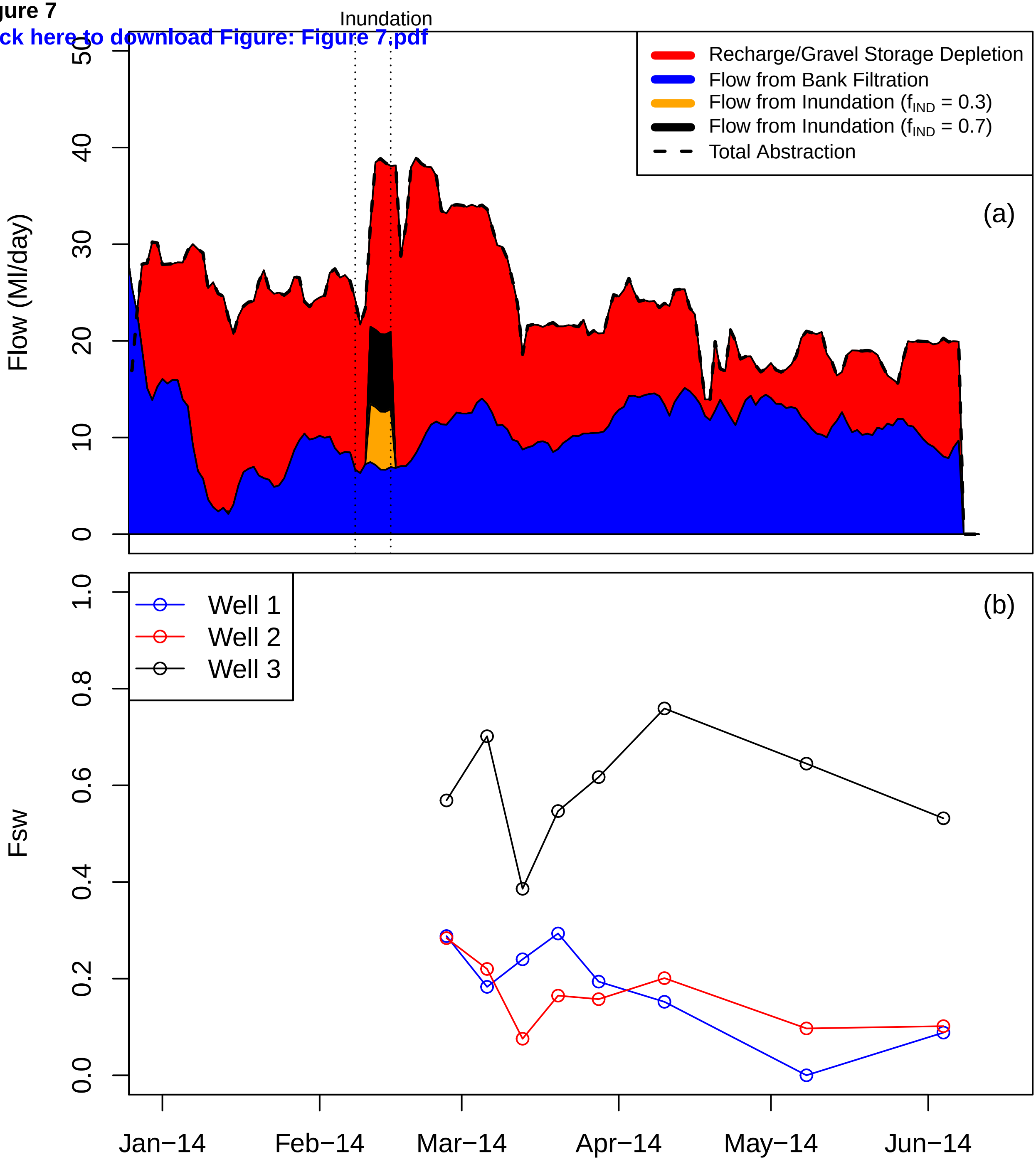

Figure 8

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