Ecological Engineering xxx (2012) xxx–xxx

Contents lists available at SciVerse ScienceDirect

Ecological Engineering

journal homepage: www.elsevier.com/locate/ecoleng

Oxygen transfer and consumption in subsurface flow treatment wetlands

Jaime Nivala[a,b,*], Scott Wallace[c], Tom Headley[d], Kinfe Kassa[b,e], Hans Brix[a], Manfred van Afferden[b], Roland Müller[b]

[a] *Department of Bioscience, Plant Biology, Aarhus University, Ole Worms Allé 1, Building 1135, 8000 Aarhus C, Denmark*
[b] *Helmholtz Center for Environmental Research (UFZ), Environmental and Biotechnology Center (UBZ), Permoserstrasse 15, 04318 Leipzig, Germany*
[c] *Naturally Wallace Consulting LLC, P.O. Box 2236, 109 E. Myrtle Street, Stillwater, MN 55082, USA*
[d] *Bauer Nimr LLC, P.O. Box 1186, Al Mina, Muscat, Oman*
[e] *TU Berlin, FG Siedlungswasserwirtschaft, Sekr. TIB 1B 16, Gustav-Meyer-Allee 25, 13355 Berlin, Germany*

ARTICLE INFO

Article history:
Received 12 April 2012
Received in revised form 19 July 2012
Accepted 10 August 2012
Available online xxx

Keywords:
Aeration
Constructed wetland
Design
Domestic wastewater
Horizontal flow
Oxygen usage
Reciprocating
Tidal flow
Vertical flow

ABSTRACT

Subsurface oxygen availability tends to be one of the main rate-limiting factors for removal of carbonaceous and nitrogenous compounds in subsurface flow (SSF) wetlands used for domestic wastewater treatment. This paper reviews the pertinent literature regarding oxygen transfer and consumption in subsurface flow treatment wetlands, and discusses the factors that influence oxygen availability.

We also provide first results from a pilot-scale research facility in Langenreichenbach, Germany (15 individual systems of various designs, both with and without plants). Based on the approach given in Kadlec and Wallace (2009), areal-based oxygen consumption rates for horizontal flow systems were estimated to be between 0.5 and 12.9 g/m^2-d; for vertical flow systems between 7.9 and 58.6 g/m^2-d; and for intensified systems between 10.9 and 87.5 g/m^2-d. In general, as the level of intensification increases, so does subsurface oxygen availability. The use of water or air pumps can result in systems with smaller area requirements (and better treatment performance), but it comes at the cost of increased electricity inputs. As the treatment wetland technology envelope expands, so must methods to compare oxygen consumption rates of traditional and intensified SSF treatment wetland designs.

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1. Introduction

Subsurface-flow treatment wetlands are commonly used for the decentralized treatment of domestic wastewater prior to soil dispersal, irrigation reuse or surface water discharge (Kadlec and Wallace, 2009). Compared to conventional wastewater treatment technologies, treatment wetlands offer many advantages: they are low-cost, robust, simple to operate, and can be constructed out of locally available materials (Wallace and Knight, 2006). These factors lend to the widespread use and implementation of treatment wetlands in areas for which centralized sewage treatment is not a cost-effective option.

Aerobic conditions allow effective removal of many common wastewater constituents such as biochemical oxygen demand (BOD), chemical oxygen demand (COD), and ammonium–nitrogen (Metcalf and Eddy Inc., 2003). In subsurface flow wetlands used for wastewater treatment, the oxygen demand exerted by the incoming wastewater generally exceeds the amount of oxygen available within the system (Kadlec and Wallace, 2009). As a result, oxygen transfer tends to be one of the main rate-limiting processes in subsurface-flow treatment wetlands.

Subsurface flow wetlands can be considered functionally similar to attached-growth bioreactors, with much of the pollutant degradation processes being undertaken by biofilms growing on the surface of the wetland substrate. Thus, for oxygen to be available for treatment processes, it can either be transferred to the water itself or to the biofilm surfaces. The prominent pathways of oxygen transfer in subsurface flow treatment wetlands are atmospheric diffusion, plant-mediated oxygen transfer, and convective flow of air within the pore space of the media (Brix and Schierup, 1990; Tanner and Kadlec, 2003; Kadlec and Wallace, 2009).

This paper reviews the mechanisms of oxygen transfer and consumption in treatment wetlands and provides an overview of the methods used to estimate oxygen transfer rates in these treatment systems. We also summarize the reported rates for commonly implemented treatment wetland designs. The findings from the review are then compared against new results from a pilot-scale facility in Germany that is comprised of 15 individual

* Corresponding author at: Helmholtz Center for Environmental Research (UFZ), Environmental and Biotechnology Center (UBZ), Permoserstrasse 15, 04318 Leipzig, Germany.
E-mail address: jaime.nivala@gmail.com (J. Nivala).

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http://dx.doi.org/10.1016/j.ecoleng.2012.08.028

wetland treatment systems. The main objective of the study was to investigate oxygen consumption rates of the various treatment wetland designs (horizontal flow, vertical flow, and intensified designs). The pilot-scale systems in Germany received the same primary-treated wastewater, enabling for the first time a true side-by-side comparison of various wetland designs. Planted and unplanted replicates were constructed in order to elucidate the role that wetland plants (*Phragmites australis*) play in oxygen transfer. Areal and volumetric oxygen consumption rates from the pilot-scale treatment systems are presented, and the limitations of current methods are discussed.

1.1. Atmospheric diffusion

Compared to free water surface (FWS) treatment wetlands, the actual surface area of the air-water interface in SSF wetlands is reduced by at least 60% due to the presence of the sand or gravel substrate. Mechanisms such as wave action and wind-induced mixing that contribute to surface reaeration in FWS wetlands are not operable in SSF wetlands; therefore atmospheric diffusion is the primary means of gas transfer. Atmospheric diffusion is further impeded by the fact that air generally must travel through a layer of unsaturated gravel and leaf litter before reaching the water surface. Because the rate of diffusion of oxygen is orders of magnitude slower through water than through air (Brix, 1993), passive diffusion processes are unlikely to have a significant impact on oxygen availability in conventional horizontal subsurface flow wetlands. Oxygen diffusion depends on various environmental factors such as water and air temperature, and degree of saturation of the bed. Tanner and Kadlec (2003) estimate that atmospheric diffusion of oxygen into a subsurface flow wetland system is on the order of 0.11 g/m^2-d, which for domestic wastewater treatment wetlands is an order of magnitude smaller than the oxygen demand of the incoming wastewater. While diffusion rates in conventional HSSF systems are quite low, diffusion can be significant in other types of treatment wetland designs, such as unsaturated vertical flow systems (Schwager and Boller, 1997).

1.2. Plant-mediated oxygen transfer

The role of plant-mediated oxygen transfer in treatment wetlands is one of the most highly debated topics in the literature. Internal transport of oxygen in wetland plants can occur via passive diffusion or through convective flow of air through plant aerenchyma (Brix et al., 1992; Brix, 1993). Oxygen release rates vary with plant species and season (Stein and Hook, 2005), as well as with the oxygen demand of the surrounding environment (Sorrell and Armstrong, 1994). In strongly reducing (e.g., wastewater) environments, wetland plants tend to minimize oxygen loss to the rhizosphere, which may limit the amount of oxygen released to growing root tips (Armstrong et al., 1990).

Some studies have aimed to directly measure plant-mediated oxygen transfer rates in SSF wetlands, while others have inferred oxygen transfer rates from water quality data. Reported rates of plant-mediated oxygen transfer in treatment wetlands span almost four orders of magnitude from 0.005 to 12 g/m^2-d (Table 1). Part of the variability in reported rates is due to differences in measurement techniques and the overall difficulty associated with measuring the oxygen concentrations at the root surface (Sorrell and Armstrong, 1994; Kadlec and Knight, 1996; Brix, 1997). There are also difficulties associated with extrapolating laboratory measurements to full-scale applications due to issues of scale and the non-homogeneity of root oxygen release (Brix, 1993).

Plant-mediated oxygen transfer in early Root Zone Method (RZM) systems was implied to be in the range of 5–25 g/m^2-d (Brix and Schierup, 1990). Similarly, a number of studies have measured oxygen consumption (based on an assumed stoichiometry for pollutant removal), and attributed that oxygen consumption entirely to plant-mediated oxygen transfer (Burgoon et al., 1989; Gersberg et al., 1989; McGechan et al., 2005). The general conclusion, however, is that the actual rates of plant-mediated oxygen transfer are not large enough to meet the demand exerted by primary treated domestic wastewater under common loading conditions (Brix and Schierup, 1990; Tanner and Kadlec, 2003; Bezbaruah and Zhang, 2005). As a result, many wetland design guidelines now neglect plant-mediated oxygen transfer altogether (U.S. EPA, 2000; Wallace and Knight, 2006; Kadlec and Wallace, 2009). Nevertheless, root release of oxygen and/or carbon compounds has been reported to affect microbial activity in SSF treatment wetlands (Zhu and Sikora, 1995; Gagnon et al., 2007; Faulwetter et al., 2009; Wu et al., 2011a). Such information suggests that while the rate of plant-mediated oxygen transfer may not be high enough to realistically meet the full oxygen demand of the wastewater, plants and/or root release of oxygen may indirectly affect treatment processes by changing the microbial community within the wetland bed (Dan et al., 2011).

1.3. Oxygen transfer at the water–biofilm interface

The limited oxygen transfer capability of conventional horizontal subsurface flow wetland designs has led to the development of alternative design configurations that improve the oxygen transfer to the subsurface zone (Brix and Schierup, 1990). These design configurations aim to provide sufficient oxygen for nitrification and removal of organic matter through use of shallow bed depth, intermittent dosing with vertical unsaturated flow, frequent water level fluctuation, or direct mechanical aeration of the gravel substratum. Although these "intensified" designs are gaining increased attention in the literature and in engineering practice, design standards for many of these types of wetlands have yet to be published (Kadlec and Wallace, 2009).

Early horizontal subsurface flow (HSSF) wetland designs were based on the Root Zone Method (RZM) (Kickuth, 1981). As discussed previously, plant-mediated oxygen transfer was thought to be a key mechanism in RZM designs, but actual oxygen transfer rates generally did not meet these design expectations (Brix, 1990) and the systems often clogged. This led to the development of vertical flow (VF) wetlands in the late 1980s (Brix and Schierup, 1990; Burka and Lawrence, 1990; Liénard et al., 1990), although the basic concept of these vertical flow wetlands goes back to the Max Planck Institute Process (MPIP) of Seidel (1966) and is similar to that of intermittent sand filters which have been in use for over 100 years (Crites and Tchobanoglous, 1998). These VF wetlands are intermittently pulse-loaded, and wastewater percolates through the unsaturated substrate. Ventilation pipes connecting a network of perforated drainage pipes to the atmosphere are often installed to provide a pathway for air to be drawn into the substrate from the bottom of the bed. Thus, air has an opportunity to enter the bed from either the top or the bottom and contact the biofilm between each loading event. This approach provides significant improvement of subsurface oxygen availability compared to HSSF designs. However, if VF wetlands are hydraulically or organically overloaded, ponding of wastewater occurs. This effectively cuts off air circulation and promotes clogging, which dramatically reduces oxygen transfer (Platzer and Mauch, 1997).

Based on hydraulic studies of typical HSSF wetlands, water was observed to bypass treatment by flowing under, as opposed to through, the plant root zone (Fisher, 1990; Breen and Chick, 1995; Rash and Liehr, 1999). García et al. (2005) investigated the treatment performance of side-by-side wetlands, some with a depth

Table 1
Reported plant oxygen release rates.

Plant	Plant oxygen release rate (g/m^2-d)	Approach	Source
Phragmites sp.	0.014–0.015	Measurement of root respiration	Ye et al. (2012)
Scirpus sp.	0.005–0.011[a]	Measurement of root respiration	Bezbaruah and Zhang (2005)
Phragmites sp.	0.02	Measurement of root respiration	Brix and Schierup (1990)
Typha sp.	0.023	Measurement of root respiration	Wu et al. (2001)
Potamogeton sp.	0.4–0.5[a]	Measurement of root respiration	Kemp and Murray (1986)
Typha sp.	0.45	Model simulation	Mburu et al. (2012)
Schoenoplectus sp.	0.94		
Carex sp.	1.91		
Phragmites sp.	1.6–3.1[b]	Measurement of root respiration	Gries et al. (1990)
Phragmites sp.	4.1	Isotope analysis	Wu et al. (2011a)
Phragmites sp.	5.0–12	Measurement of root respiration	Armstrong et al. (1990)

[a] Originally reported in mg/m^2-d.
[b] Originally reported in mg/m^2-h.

typical of most HSSF designs (50 cm) and some with a water depth limited to the rooting depth of the plants (in their study, it was 27 cm). Their results showed greater removal of organic matter and ammonia nitrogen in the shallow beds. Such findings indicate that oxygen transfer into HSSF wetlands by diffusion or root-oxygen release can be optimised without any additional energy inputs by simply limiting the wetted depth of the bed to the depth of the plant roots (generally 25–30 cm).

Another means of drawing air into a wetland bed is through the sequential filling and draining of the wastewater through the wetland substrate. As the wetland bed is drained, air is drawn into the bed (Green et al., 1997), oxygenating the exposed biofilms on the wetland substratum. This improves treatment performance compared to systems with a static water level (Tanner et al., 1999; Liebowitz et al., 2000). Since the rate of air circulation (and thus oxygen transfer) is related to the frequency of water level fluctuation, internal recycling to rapidly fill and drain multiple wetland compartments is often employed (Behrends et al., 1996; Austin et al., 2003; Ronen and Wallace, 2010). These systems are commonly termed “reciprocating”, “tidal flow” or “fill-and-drain” wetlands.

Mechanical aeration of SSF wetlands using air distribution pipes installed at the bottom of the wetland bed has also been utilized as a means to increase oxygen transfer in wetland treatment systems. This includes aeration of HSSF wetlands (Wallace, 2001; Higgins, 2003; Ouellet-Plamondon et al., 2006; Maltais-Landry et al., 2009) and saturated VF wetlands (Murphy and Cooper, 2011; Wallace and Liner, 2011).

2. Methods for estimating oxygen transfer and consumption rates

Historically, oxygen usage rates have been inferred from water quality data based on the amount of oxygen-consuming pollutants removed by the wetland. This approach requires the use of stoichiometry and influent–effluent water quality data (Schwager and Boller, 1997; Liénard et al., 1998; Cooper, 2005). Equations used for estimating oxygen usage vary widely in the literature. We note a distinction between the commonly used term: *oxygen transfer rate* and what we consider to be a more technically accurate term: *oxygen consumption rate*. The term *oxygen transfer* implies quantification of the total amount of oxygen that has physically passed into the subsurface wetland environment, whereas estimates based on inlet and outlet water quality data actually estimate the net amount of oxygen *consumed* in a system. Due to the dynamic and complex nature of the simultaneous processes in treatment wetlands, historically it has not been possible to directly quantify oxygen transfer rates. As such, we recommend the term *oxygen consumption rate* when using water quality data to infer how much oxygen has been consumed in a particular wetland treatment system. Oxygen transfer represents the upper limit of the potential oxygen consumption, and is thus an extremely important design parameter for treatment wetland systems.

In recent years, new methods have been developed to better quantify in situ oxygen transfer and oxygen consumption in treatment wetlands. These methods are still under development and refinement, and include the gas tracer method (measuring oxygen transfer), the respirometry method (estimating oxygen consumption) and inference from water quality data (estimating oxygen consumption).

2.1. Gas tracer method

The use of an inert gas as a tracer has been used to estimate oxygen transfer rates in wastewater treatment technologies such as rotating biological contactors (Boumansour and Vasel, 1998). Schwager and Boller (1997) applied sulphur hexafluoride (SF_6) to monitor enclosed air in an unsaturated vertical flow wetland, verifying that molecular gas diffusion is the dominant process responsible for high oxygen transfer rates in this type of system. With this method, they estimated a maximum oxygen flux of 55 g/m^2-d.

Santa (2007) used propane (C_3H_8) to estimate an oxygen transfer rate for a laboratory scale unplanted HSSF wetland system with 48 h retention time, reporting a rate of 0.78 g O_2/m^2-d. Tyroller et al. (2010) further investigated the suitability of propane gas (87.5% purity) on a small (0.7 m^2) HSSF system planted with *P. australis*. They reported an inverse correlation between hydraulic retention time and oxygen transfer rates, probably related to turbulence at the relatively short retention times they investigated. Oxygen transfer rates of 2.3–3.2 g O_2/m^2-d were reported for a 15-h hydraulic retention time, and 0.2–0.6 g O_2/m^2-d for a hydraulic retention time of 45 h.

2.2. Respirometry methods

The respirometry method is derived from conventional activated sludge wastewater treatment technology (Spanjers and Vangrolleghem, 1995). It enables the quantification of microbial respiration, from which kinetic parameters can be derived. Andreottola et al. (2007) modified the approach to fit a lab-scale vertical flow wetland column. Their approach enabled the elucidation of various components of microbial oxygen consumption in a saturated water column, including: endogenous respiration, nitrification, and oxidation of readily and slowly degradable

COD. Further work by Ortigara et al. (2010) using this same approach reported a maximum areal-based oxygen usage rate of 73 g O_2/m^2-d for unplanted saturated vertical flow lab-scale mesocosms fed with wastewater.

Morvannou et al. (2010, 2011) developed a solid respirometry method in order to assess microbial activity in an unsaturated vertical flow wetland. The solid respirometry approach resulted in a higher reported maximum rate of nitrification (41.3 g O_2/m^3-h) compared to the liquid respirometry approach of Ortigara et al. (2010) (1.8 g O_2/m^3-h). The difference in results between the two methods is attributed to how oxygen was measured (e.g., water phase vs. air phase) (Morvannou et al., 2010). To date, respirometry methods have not been applied to full-scale wetland cores. However, these methods have been identified as a promising tool for better understanding the mechanisms involved in oxidation of carbonaceous and nitrogenous compounds in treatment wetlands (Langergraber, 2010).

2.3. Inference from water quality data

The common approach for estimating oxygen usage in treatment wetlands is through the use of stoichiometric relationships, water quality data, and basic assumptions about how pollutants are degraded in a wetland system. Each calculation includes two main components: first, an estimate of the carbonaceous biochemical oxygen demand removed in the wetland; and second, an estimate of nitrogenous biochemical oxygen demand removed. The implied oxygen consumption is then usually defined as the sum of the carbonaceous and nitrogenous components, and is generally reported as a rate that has been normalized to the surface area of the wetland bed (g/m^2-d). Eq. (1) (Liénard et al., 1998), Eq. (2) (Platzer, 1999), and Eq. (3) (Cooper, 2005) show some of the various ways oxygen consumption rate (OCR) has been estimated in the treatment wetland literature.

$$\mathrm{OCR} = \frac{[1.0(\Delta M_{\mathrm{COD}}) + 4.5(\Delta M_{\mathrm{TKN}})]}{A} \quad (1)$$

$$\mathrm{OCR} = \frac{[0.7(\Delta M_{\mathrm{COD}}) + 4.3(\Delta M_{\mathrm{TKN}}) - 2.9(\Delta M_{\mathrm{NO_3\text{-}N}})]}{A} \quad (2)$$

$$\mathrm{OCR} = \frac{[1.0(\Delta M_{\mathrm{BOD_5}}) + 4.3(\Delta M_{\mathrm{NH_4\text{-}N}})]}{A} \quad (3)$$

where ΔM is the mass removed for a specific parameter ($= Q_i(C_i - C_o)$, g/d); Q_i is the inflow, m^3/d; C_i is the inlet concentration, mg/L = g/m^3; C_o is the outlet concentration, mg/L = g/m^3; A is the area, m^2.

Simple inspection of Eqs. (1)–(3) show the differences between estimation methods. The difference between Chemical Oxygen Demand (COD), 5-day Biochemical Oxygen Demand (BOD_5) and 5-day Carbonaceous Biochemical Oxygen Demand ($CBOD_5$) are typically not well distinguished in the wetland literature (Table 2). The use of COD measures both biodegradable and non-biodegradable components; for the purpose of estimating oxygen usage, the use of COD will result in overestimated oxygen consumption since not all of the removed COD is necessarily aerobically biodegraded. When calculating oxygen usage rates, a factor of 0.7 is generally applied to COD values to estimate the biodegradable component of COD (Platzer, 1999). The BOD_5 and $CBOD_5$ tests both measure the oxygen required for decomposition of the bioavailable carbonaceous fraction. Conventional practice is to run the test for five days, but in some studies the test is run for seven days instead (Mander et al., 2003). The BOD_5 test measures the oxygen consumed by the decomposition of biodegradable carbonaceous material *and* ammonia nitrogen. The $CBOD_5$ test is conducted with a chemical that inhibits nitrification. The use of a nitrification inhibitor enables measurement of the carbonaceous component only. It is important to note that the treatment wetland literature generally uses the terms BOD_5 and $CBOD_5$ interchangeably creating a potential risk when comparing results from one study to those of another.

Table 2
Common measurements used to characterize oxygen demand in wastewater.

Measurement and abbreviation	Description
Chemical Oxygen Demand (COD)	Oxygen is consumed by decomposition of both biodegradable and non-biodegradable organic material. Ammonia is not oxidized in this test.
5-day Biochemical Oxygen Demand (BOD_5)	Oxygen is consumed by decomposition of both biodegradable organic material and ammonia nitrogen. The test is run for five days.
5-day Carbonaceous Biochemical Oxygen Demand ($CBOD_5$)	Oxygen is consumed by decomposition of biodegradable organic material only. Microbial nitrification is chemically inhibited. The test is run for five days.

The next factor to consider is carbon degradation pathways in treatment wetlands. The amount of organic matter degraded aerobically (vs. anaerobically) is generally unknown for a given wetland system. In horizontal subsurface flow wetlands, the oxygen demand applied often exceeds the rate of oxygen transfer into the system, thus anaerobic pathways become an important mechanism for removal of organic matter. Overall removal (without distinction between pathways) is typically reported, although anaerobic, anoxic, and aerobic mechanisms are all potentially important as noted in Brix (1990) and Tanner et al. (1999). Tanner and Kadlec (2003) note that most studies estimating oxygen usage from water quality data have assumed that all BOD removal occurs via aerobic processes, which is likely to result in an over-estimate of oxygen consumption in a wetland treatment system. Brix (1990) took a carbon and oxygen mass balance approach and found that 9% of the organic loading was deposited or decomposed anaerobically in the bed. Ojeda et al. (2008) investigated the relative importance of anaerobic vs. anoxic/aerobic COD degradation pathways in a 2D simulation model, and suggest that between 60% and 70% of organic matter degradation could be attributed to anaerobic removal pathways (particularly methanogenesis and sulphate reduction). The results from another simulation study by Llorens et al., 2011 concur with the values reported by Ojeda et al. (2008), citing 71.85–78.88% of organic matter removal occurring via anaerobic pathways. They suggest that methanogenesis and sulphate reduction can occur within the wetland simultaneously, and potentially at the same locations. Similarly, Bezbaruah and Zhang (2009) estimated that 64% of BOD is degraded through aerobic routes, and the remaining 36% is degraded anaerobically (although nitrogen cycle and temperature were not considered in their study). The uncertainty over the spatial and temporal overlap of various carbon degradation pathways has an impact when estimating oxygen consumption in treatment wetland systems. Oxygen consumption estimates based on water quality data alone do not, for example, account for the retention and accumulation of particulate organic matter within a system. For these reasons, Kadlec and Wallace (2009) propose an approach where a range of OCRs should be calculated, including a minimum scenario where all $CBOD_5$ removal is assumed to occur via anaerobic pathways or particulate accumulation.

There is a similar level of ambiguity when estimating the nitrogenous component of the oxygen usage calculation. The most common method is to base the estimation on the decrease in ammonia concentration in the system (Cooper, 1999; Platzer, 1999; Noorvee et al., 2005a). However, this approach does not take into consideration biological transformation of organic nitrogen to ammonia (ammonification) and subsequent nitrification of that ammonia. It is therefore recommended to use Total Kjeldahl Nitrogen (TKN) concentrations in lieu of ammonium nitrogen concentrations when calculating oxygen consumption rates in treatment wetlands.

It is also worth reviewing the assumptions surrounding ammonium removal in treatment wetlands. Tanner and Kadlec (2003) point out that most studies assume ammonia removal occurs via the classical nitrification–denitrification sequence. However, alternate nitrogen removal pathways such as anaerobic ammonia oxidation (Anammox) (Jetten et al., 1999) have called into question the appropriateness of this assumption. With recent advances in microbial identification and quantification methods, bacteria involved in alternate nitrogen removal pathways have been reported in various types of treatment wetlands (Austin et al., 2003; Shipin et al., 2004; Dong and Sun, 2007; Paredes et al., 2007; Tao et al., 2011) and other studies have claimed alternate nitrogen pathways from stoichiometry-based analyses (Tanner and Kadlec, 2003; Bishay and Kadlec, 2005; Sun and Austin, 2007). Zhu et al. (2010) emphasize the potential importance of alternate nitrogen removal pathways especially in large-scale, nitrogen-rich wetland applications. Oxidation of ammonia through the use of nitrite results in a much lower oxygen requirement compared to conventional nitrification (1.94 g O/g NH_4-N vs. 4.57 g O/g NH_4-N) (Kadlec and Wallace, 2009), but the fraction of nitrogen removed through such pathways is unknown.

Despite the limitations of using water quality data, this method is commonly employed since other methods (gas tracer, respirometry) require specialized equipment and are confined to laboratory-scale experiments. In the case of full-scale treatment wetlands, water quality data are generally the only information available to estimate oxygen consumption. Kadlec and Wallace (2009) suggest an approach which takes into consideration the spectrum of uncertainties regarding carbon and nitrogen oxidation in treatment wetlands. They estimate oxygen consumption rates (OCR) with three equations (Eqs. (4)–(6)) for the maximum, intermediate, and minimum stoichiometric cases. The nitrogenous demand is calculated using Total Kjeldahl Nitrogen, in order to account for organic nitrogen that may be ammonified and thus, contribute an internal ammonia nitrogen load to the system. For the maximum case, a TKN stoichiometric coefficient of 4.6 is chosen (reflecting conventional nitrification); for the intermediate and minimum case estimates, a TKN stoichiometric coefficient of 1.7 is chosen (to reflect alternative nitrogen pathways). The carbonaceous component is calculated from $CBOD_5$, with a stoichiometric coefficient of 1.5 for the maximum case, 1.0 for the intermediate case, and zero for the minimum (e.g., assuming that all $CBOD_5$ is removed anaerobically). Because inflow and outflow data are rarely available, the average inflow rate is typically used in these calculations. However, mass removals should be calculated from inflow and outflow values when such data are available.

$$\mathrm{OCR}_{\mathrm{Maximum}} = \frac{[1.5(\Delta M_{\mathrm{CBOD_5}}) + 4.6(\Delta M_{\mathrm{TKN}})]}{A} \quad (4)$$

$$\mathrm{OCR}_{\mathrm{Intermediate}} = \frac{[1.0(\Delta M_{\mathrm{CBOD_5}}) + 1.7(\Delta M_{\mathrm{TKN}})]}{A} \quad (5)$$

$$\mathrm{OCR}_{\mathrm{Minimum}} = \frac{[1.7(\Delta M_{\mathrm{TKN}})]}{A} \quad (6)$$

where ΔM is the mass removed for a specific parameter(=$Q_iC_i - Q_oC_o$), g/d; Q_i is the inflow, m^3/d; Q_o is the outflow, m^3 d; C_i is the inlet concentration, mg/L = g/m^3; C_o is the outlet concentration, mg/L = g/m^3; A is the area, m^2.

Reported oxygen consumption rates from the treatment wetland literature are summarized in Table 3. Since the distinction between oxygen consumption and oxygen transfer is not often made in the wetland literature, many of the previous studies in Table 3 have reported results as an oxygen transfer rate, when in fact, the calculated result was an oxygen consumption rate. Most values have been estimated according to water quality data. Reported oxygen consumption rates for horizontal flow wetlands are generally lower than 10 g/m^2-d, whereas reported rates for vertical flow wetlands are nearly an order of magnitude higher. Rates for intensified wetlands or hybrid (combination) systems are yet higher; although care should be exercised in extrapolating results from highly controlled laboratory environments to full-scale designs.

3. Research facility in Langenreichenbach, Germany

A pilot-scale research facility near the village of Langenreichenbach, Germany was commissioned in 2010 to compare the relative merits and capabilities of conventional and alternative ecotechnologies, with a specific focus on design configurations that overcome the limitation of subsurface oxygen availability. Planted and unplanted replicates were constructed in order to elucidate the role that wetland plants (*P. australis*, among others) play in oxygen transfer. The site contains 15 individual treatment systems, which are briefly described in Table 4. All systems are loaded with municipal wastewater that has first been passed through a sedimentation tank for primary treatment. For a detailed description of the overall research facility and each specific design, the reader is referred to Nivala et al. (in preparation).

Table 5 summarizes the influent and effluent water quality data for the 15 treatment systems. Each system was operated at its design loading throughout the course of this study. Table 6 provides oxygen consumption rates for each system based on inflow and outflow rates and water quality data as outlined in the previous section (Eqs. (4)–(6)). Using Tables 5 and 6, comparisons can be made between the different wetland configurations (HSSF, VF and intensified), and how the Langenreichenbach wetlands compare to systems reported in the literature.

The horizontal flow systems (H25, H25p, H50 and H50p) removed $CBOD_5$ and TN; however effluent ammonia concentrations were at or around influent concentrations due to internal ammonification of organic nitrogen. The lack of significant ammonia removal and relatively high effluent CBOD concentrations indicates that the treatment environment was oxygen limited, thus the range of oxygen consumption rates (OCR) reported in Table 6 (0.4–12.9 g/m^2-d; depending on stoichiometric assumptions) is likely to bracket the range of OCR rates possible in a HSSF wetland system. These results also closely match the inferred oxygen consumption rates reported by Kadlec and Wallace (2009) from their aggregated data set (approximately 362 HSSF wetlands).

The deeper HF systems (H50 and H50p) had higher areal OCRs than the shallow systems (H25 and H25p), which is not consistent with the data presented by García et al. (2004). However, both H50 and H50p were loaded at 0.20 m^3/d (double the loading rate of H25 and H25p). While the shallow beds (H25 and H25p) delivered lower effluent concentrations, the load removed in the deeper beds was higher due to the higher flow rate applied to H50 and H50p, thereby resulting in higher areal OCRs. Looking at the difference between planted and unplanted beds (H25 vs. H25p and H50 vs. H50p), the planted beds had higher oxygen consumption rates, indicating that vegetation (in this study, *P. australis*) enhanced the transfer of oxygen to the substrate. Most interestingly, the shallow beds (H25 and H25p) showed the greatest effect of vegetation on OCR, presumably

Table 3
Reported oxygen consumption rates for HSSF, VF, intensified, and hybrid treatment wetland systems.

System	Scale	Method used for estimation[a]	Oxygen consumption rate (g/m^2-d)	Source
Horizontal flow				
HF	Laboratory	Gas tracer	0.3–5.0	Tyroller et al. (2010)
HF	Laboratory	Gas tracer	0.78	Santa (2007)
HF	Full	Water quality data (BOD, NH_4)	2.43	McGechan et al. (2005)
HF	Full	Water quality data (BOD, NH_4)	2.7	Noorvee et al. (2005b)
HF	Full	Water quality data (BOD, NH_4)	3.87–11.6	Gasiunas (2011)
HF	Full	Water quality data (BOD, NH_4)	5.5–10.0	Gersberg et al. (1986)
HF	Full	Water quality data (BOD, NH_4)	7.3	Headley et al. (2005)
Vertical flow				
VF	Laboratory	Gas tracer	56	Schwager and Boller (1997)
VF	Laboratory	Respirometry	49	Andreottola et al. (2007)
VF	Laboratory	Respirometry	73	Ortigara et al. (2010)
VF	Laboratory	Water quality data (BOD, NO_x)	29.7–57.1	Sun et al. (2002)
VF	Laboratory	Water quality data (BOD, NH_4)	147–156	Ye et al. (2012)
VF	Laboratory	Water quality data (COD, NH_4)	60–80	Kantawanichkul et al. (2009)
VF	Full	Water quality data (BOD, NH_4)	5.7–18.4	Gasiunas (2011)
VF	Full	Water quality data (COD, TKN, NO_3)	28.4–35.4	Weedon (2003)
VF	Full	Water quality data (COD, TKN)	55	Kayser and Kunst (2005)
VF	Full	Water quality data (BOD, NH_4)	63.6	Noorvee et al. (2005b)
VF	Full	Water quality data (BOD, NH_4)	92	Johansen et al. (2002)
VF	n/a	Numerical simulation	13.6	Petitjean et al. (2012)
VF (French)	n/a	Numerical simulation	90	Petitjean et al. (2012)
VF (French)	Full	Water quality data (COD, TKN)	68	Liénard et al. (1998)
Intensified or hybrid systems				
Hybrid	Full	Water quality data (BOD, NH_4)	40–79	Cooper (2003)
HF + aeration	Full	Water quality data (BOD, NH_4)	50–100	Kadlec and Wallace (2009)
HF + aeration	Full	Water quality data (BOD, NH_4)	134	Wallace (2002)
Tidal flow	Laboratory	Gaseous O_2 measurements	350	Wu et al. (2011b)
Tidal flow	Laboratory	Water quality data (BOD, NH_4)	482	Sun et al. (2005)
Tidal flow	Full	Water quality data (BOD, NH_4)	30	Cooper and Cooper (2005)
VF + batch loading	Pilot	Water quality data (BOD, NH_4)	21.1	Karabelnik et al. (2008)
VF + recirculation	Full	Water quality data (BOD, NH_4)	87	Noorvee (2007)
VF + passive air pump	Laboratory	Water quality data (NH_4)	30–80	Green et al. (1998)
VF + passive air pump	Laboratory	Water quality data (NH_4)	520–4760	Lahav et al. (2001)
VF + aeration	Full	Water quality data (BOD, NH_4)	48	Murphy and Cooper (2011)
VF + aeration	Full	Water quality data (BOD)	1027	Wallace and Liner (2011)

[a] Gas tracer experiments measure the rate of oxygen transfer to the subsurface.

Table 4
Design and operational details of the 15 treatment systems at Langenreichenbach, Germany.

System abbreviation[a]	System type	Effective depth[b] (cm)	Saturation status	Main media	Hydraulic loading rate (L/m^2-d)
H25, H25p	HF	25	Saturated	8–16 mm gravel	18
H50, H50p	HF	50	Saturated	8–16 mm gravel	36
VS1, VS1p[c]	VF	85	Unsaturated	1–3 mm sand	95
VS2, VS2p[d]	VF	85	Unsaturated	1–3 mm sand	95
VG, VGp	VF	85	Unsaturated	4–8 mm gravel	95
VA, VAp	VF + aeration	85	Saturated	8–16 mm gravel	95
HA, HAp	HF + aeration	100	Saturated	8–16 mm gravel	130
R	Reciprocating	95	Alternating	8–16 mm gravel	160

[a] Systems planted with *Phragmites australis* are denoted with "p" in the system abbreviation.
[b] Effective depth refers to the depth of the media involved in treatment. Depth of media not involved in treatment (such as the fill above distribution shields in a vertical flow bed, or the layer of dry gravel in a saturated bed) is not considered.
[c] Systems were dosed once every hour.
[d] Systems were dosed once every 2 h.

because the plant rhizosphere was able to occupy a greater portion of the overall bed volume.

The OCRs of the unsaturated VF systems were more than double the mean values reported by Kadlec and Wallace (2009). This makes sense, because the Langenreichenbach VF systems were operated at a higher hydraulic loading rate (95 mm/d) compared to those in the dataset of Kadlec and Wallace (43 mm/d). The sand-based systems (VS1, VS1p, VS2, VS2p) displayed higher OCRs than the gravel-based systems (VG, VGp), indicating that the size of the media plays an important role in the overall treatment process. Interestingly, the gravel-based vertical flow systems had higher total N removal rates, possibly because the lower oxygen availability provided more favorable conditions for denitrification compared to the sand-based beds. Concurrent denitrification would mean that some of the observed CBOD removal was consumed anaerobically by denitrifying bacteria, therefore leading to a slight overestimate for the maximum OCR (Eq. (6)); however within the range of stoichiometric assumptions presented in (Eqs. (4)–(6)), the calculated range of OCRs are still valid. The difference in dosing frequency used for VS1 and VS1p (every hour) vs. VS2 and VS2p (every 2 h) did not appear to play a role in treatment efficiency, indicating that net effect of these dosing regimes did not have a large impact on OCR. For the sand-based VF systems (VS1, VS1p, VS2, VS2p), vegetated systems showed slightly higher OCR values than non-planted systems (similar to the HF results), indicating that plants may slightly improve inferred OCR. However, it is worth noting that

Table 5
Water quality data for the 15 treatment wetland systems at Langenreichenbach, Germany.

System[a]	Area (m^2)	Effective depth (m)	Flow		$CBOD_5$		TN		NH_4-N		TKN[e]		NO_x-N[e]		Org-N[e]	
			Q_i (m^3/d)	Q_o (m^3/d)	C_i (mg/L)	C_o (mg/L)	C_i (mg/L)	C_o (mg/L)	C_i (mg/L)	C_o (mg/L)	C_i (mg/L)	C_o (mg/L)	C_i (mg/L)	C_o (mg/L)	C_i (mg/L)	C_o (mg/L)
Horizontal flow[b]																
H25	5.64	0.25	0.10	0.11	236 ± 80	48.8 ± 18.5	72.8 ± 16.6	56.0 ± 12.6	54.6 ± 17.1	54.6 ± 19.1	72.4	55.7	0.4	0.3	17.8	1.1
H25p	5.64	0.25	0.10	0.09	236 ± 80	43.4 ± 19.4	72.8 ± 16.6	50.4 ± 13.6	54.6 ± 17.1	49.5 ± 20.7	72.4	50.2	0.4	0.3	17.8	0.6
H50	5.64	0.50	0.20	0.21	234 ± 78	60.1 ± 21.5	72.7 ± 16.6	57.7 ± 11.8	54.4 ± 16.7	56.7 ± 19.7	72.3	57.4	0.4	0.2	17.9	0.7
H50p	5.64	0.50	0.20	0.19	234 ± 78	66.0 ± 25.1	72.7 ± 16.6	55.8 ± 11.7	54.4 ± 16.7	52.9 ± 16.4	72.3	55.6	0.4	0.3	17.9	2.7
Vertical flow[c]																
VS1	6.20	0.85	0.58	0.58	230 ± 78	7.6 ± 10.2	72.0 ± 16.9	54.4 ± 13.5	53.2 ± 15.8	11.0 ± 12.6	71.7	13.9	0.4	40.5	18.5	2.8
VS1p	6.20	0.85	0.58	0.56	230 ± 78	3.9 ± 4.8	72.0 ± 16.9	52.1 ± 14.1	53.2 ± 15.8	6.8 ± 10.5	71.7	10.3	0.4	41.7	18.5	3.5
VS2	6.20	0.85	0.58	0.59	230 ± 78	5.3 ± 4.4	72.0 ± 16.9	58.3 ± 12.8	53.2 ± 15.8	11.2 ± 13.6	71.7	15.6	0.4	42.6	18.5	4.4
VS2p	6.20	0.85	0.58	0.55	230 ± 78	4.1 ± 3.5	72.0 ± 16.9	56.2 ± 13.0	53.2 ± 15.8	6.8 ± 10.5	71.7	10.9	0.4	45.4	18.5	4.0
VG	6.20	0.85	0.59	0.59	230 ± 78	21.5 ± 17.6	72.0 ± 16.9	47.0 ± 10.8	53.2 ± 15.8	16.8 ± 11.8	71.7	20.4	0.4	26.6	18.5	3.6
VGp	6.20	0.85	0.59	0.58	230 ± 78	31.3 ± 27.5	72.0 ± 16.9	50.0 ± 12.5	53.2 ± 15.8	18.0 ± 12.1	71.7	22.6	0.4	27.4	18.5	4.6
Intensified[d]																
VA	6.20	0.85	0.59	0.59	233 ± 76	4.0 ± 4.5	72.0 ± 17.0	39.8 ± 8.1	54.9 ± 16.6	0.9 ± 0.9	71.6	4.5	0.3	35.3	16.7	3.6
VAp	6.20	0.85	0.59	0.58	233 ± 76	5.0 ± 4.4	72.0 ± 17.0	43.3 ± 9.9	54.9 ± 16.6	0.5 ± 0.3	71.6	4.1	0.3	39.2	16.7	3.6
HA	5.64	1.00	0.74	0.74	236 ± 74	2.4 ± 3.9	72.9 ± 16.8	40.3 ± 13.6	55.9 ± 17.2	0.4 ± 0.9	72.6	4.4	0.3	35.9	16.6	3.3
HAp	5.64	1.00	0.74	0.72	236 ± 74	1.9 ± 2.9	72.9 ± 16.8	39.9 ± 13.6	55.9 ± 17.2	0.5 ± 0.6	72.6	3.5	0.3	36.4	16.6	3.0
R	13.2	0.95	1.93	1.93	206 ± 84	3.4 ± 3.8	67.4 ± 19.2	18.7 ± 9.1	49.2 ± 17.9	4.3 ± 10.5	66.9	6.9	0.5	11.8	17.7	2.6

[a] Although the influent wastewater came from a single source, average influent concentrations for each system may vary due to operation and maintenance activities and sampling schedule. Data were collected between August 2010 and December 2011. Mean values and standard deviations are presented for measured water quality parameters.
[b] Based on 40–46 sampling events.
[c] Based on 43–45 sampling events.
[d] Based on 28–34 sampling events.
[e] Calculated from mean values.

Table 6
Oxygen consumption rates observed for the 15 treatment wetland systems at Langenreichenbach, Germany.

System	Areal oxygen consumption rate[a]			Volumetric oxygen consumption rate[a,b]		
	Maximum (g/m^2-d)	Intermediate (g/m^2-d)	Minimum (g/m^2-d)	Maximum (g/m^3-d)	Intermediate (g/m^3-d)	Minimum (g/m^3-d)
Horizontal flow						
H25	6.3	3.8	0.5	25.5	15.5	1.9
H25p	7.9	4.5	0.9	31.6	18.1	3.7
H50	11.8	7.1	0.9	23.5	14.3	1.7
H50p	12.9	7.6	1.3	25.8	15.1	2.6
Kadlec and Wallace (2009) 50th percentile	*6.3*	*3.2*	*1.0*	–	–	–
Kadlec and Wallace (2009) 80th percentile	*12.8*	*7.5*	*2.0*	–	–	–
Vertical flow						
VS1	56.1	30.0	9.2	66.0	35.3	10.8
VS1p	58.4	31.0	9.8	68.7	36.5	11.5
VS2	56.0	30.2	9.0	65.9	35.5	10.5
VS2p	58.6	31.2	9.8	68.9	36.7	11.6
VG	52.0	28.0	8.2	61.2	33.0	9.7
VGp	49.8	26.8	7.9	58.6	31.5	9.3
Kadlec and Wallace (2009) 50th percentile	*24.7*	*13.4*	*3.5*	–	–	–
Kadlec and Wallace (2009) 80th percentile	*39.9*	*20.0*	*9.1*	–	–	–
Intensified[c]						
VA	62.2	32.7	10.9	73.2	38.5	12.8
VAp	62.3	32.7	11.0	73.3	38.5	12.9
HA	86.8	45.7	15.1	86.8	45.7	15.1
HAp	87.5	46.0	15.4	87.5	46.0	15.4
R	84.8	44.6	14.9	89.3	46.9	15.7

[a] Rates have been calculated using average daily inflow and outflow rates.
[b] Volumetric calculations are based on effective depth as listed in Table 4.
[c] Systems were loaded at design load. Effluent concentrations of $CBOD_5$ and ammonia for intensified systems were very low (< 5 mg/L) indicating that oxygen consumption rates would be higher (but not necessarily sustainable) with increased loading.

this increased OCR in the presence of plants was mainly due to an increased rate of NH_4-N removal, which may be a result of plant uptake rather than enhanced nitrification in the system. Plants do not seem to play a significant role in OCR for the gravel-based systems (VG and VGp).

The intensified wetland systems (HA, HAp, VA, VAp and R) easily out-performed the passive horizontal flow (H25, H25p, H50, H50p) and gravel-based VF (VG and VGp) wetland treatment systems. While the differential is smaller for the passive sand-based vertical flow systems (VS1, VS1p, VS2, VS2p), the intensified systems still achieved lower effluent concentrations, especially with regards to ammonium-N. This is reflected in the overall OCR results (Table 6) that combine the effects of contaminant removal and applied hydraulic load. Since the intensified systems had such low effluent concentrations (<5 mg/L $CBOD_5$ and <5 mg/L NH_4-N) despite the higher hydraulic loadings (Table 4), the OCR values reported here probably do not represent the maximum OCR values that can be achieved in intensified wetland systems. It should be noted, however, that while higher OCR values may be achievable with increased load, such loads may not be sustainable in the long term, especially with regards to substrate clogging. The systems in this study were operated at design flow throughout the course of the experiment.

The use of area-based rate coefficients has been advocated in the literature (Kadlec and Wallace, 2009) based on the assumption that many treatment processes in passive wetland systems (such as oxygen transfer) are proportional to the surface area of the wetland. Wetland configurations such as aerated beds and reciprocating systems challenge that assumption; indicating that volume-based rate coefficients may be a more appropriate design tool for intensified wetlands. Table 6 provides volumetric-based OCR results for the 15 systems at Langenreichenbach. It is interesting to note the differences between the areal and volumetric OCR results, particularly for the horizontal flow beds (H25, H25p, H50, H50p). On a volumetric basis, the shallow systems perform better than the deeper beds. This is especially pronounced for the planted shallow system (H25p), which has the highest volumetric OCR of the passive horizontal flow beds.

4. Conclusions

Aerobic conditions allow for effective removal of carbonaceous and nitrogenous compounds in subsurface flow treatment wetlands. Generally, the oxygen demand exerted by the incoming wastewater exceeds the amount of oxygen available within the system, rendering oxygen availability one of the main rate-limiting processes in these treatment systems. The main mechanisms for oxygen transfer in subsurface flow treatment wetlands are atmospheric diffusion, plant-mediated oxygen transfer, and oxygen transfer at the water–biofilm interface. Wetland designs today aim at improving oxygen transfer at the water–biofilm interface, and include modifications such as artificial aeration or fill-and-drain operation.

Oxygen consumption rates in treatment wetlands are most commonly inferred from water quality data. Multiple approaches are available, making it difficult to compare results from one study to the next. The approach of Kadlec and Wallace (2009) allows consideration for various carbon and nitrogen removal pathways, and offers a range of oxygen consumption estimates as opposed to a singular estimate. This approach was applied to data from 15 different treatment systems at Langenreichenbach, Germany. Areal-based oxygen consumption rates for passive horizontal flow systems were estimated to be between 0.5 and 12.9 g/m^2-d; for vertical flow systems between 7.9 and 58.6 g/m^2-d; and for intensified systems between 10.9 and 87.5 g/m^2-d. Areal-based values do not provide a fair basis of comparison across the technology gradient from passive to intensified systems. As such volumetric OCRs were also calculated, to provide a basis of comparison for treatment systems of different depths. Volumetric-based OCRs for horizontal flow systems were estimated to be between 1.7 and 31.6 g/m^3-d; for vertical flow systems between 9.3 and 68.9 g/m^2-d; and for intensified systems between 12.8 and 89.3 g/m^3-d. The shallow (25 cm deep) HSSF systems performed slightly better than the deeper (50 cm deep) when OCR was considered on a volumetric basis.

While the stoichiometry-based approach is a useful tool to compare oxygen consumption rates in different treatment wetland systems, the approach has some limitations. The stoichiometric scenarios used in calculating OCR from water quality data span a range of basic assumptions, none of which account for retention and accumulation of particulate organic matter. Anaerobic degradation of organic matter may be a significant removal process in horizontal flow wetland systems, which is why CBOD removal is not included in the calculation of minimum OCRs. OCR for systems may be overestimated when assuming the maximum case if denitrification is occurring, which reflects an anaerobic consumption of some of the CBOD. However the values still fall within the range of stoichiometric assumptions presented. Many of the intensified wetlands (and some VF wetlands) removed $CBOD_5$ and NH_4-N to very low concentrations. Thus, the oxygen transfer rates may actually be higher than the results reported for the treatment systems at Langenreichenbach. However, while higher loadings to intensified wetlands may result in higher observed consumption rates, those rates may not necessarily be sustainable over the long term operation of the system.

Acknowledgements

This work was supported by funding from the German Ministry of Education & Research (BMBF) within the context of the SMART Project (Ref. 02WM1080). Jaime Nivala acknowledges the Helmholtz Interdisciplinary Graduate School for Environmental Research (HIGRADE) and the Helmholtz Center for Environmental Research (Helmholtz Zentrum für Umweltforschung – UFZ) for additional funding and support. The authors are particularly grateful to Katy Bernhard (UFZ) for her support in the design, construction, operation, and weekly sampling of the facility at Langenreichenbach. We also thank the following UFZ staff: Grit Weichert and Petra Hoffman for their support and assistance in sample collection and analysis; Dr. Sybille Mothes, Jürgen Steffen, Carola Bönisch, and Karsten Marien for analytical support; and Thomas Aubron for his thoughtful feedback during manuscript preparation. We also acknowledge the many others who contributed to the renewal, re-commissioning, and operation of Langenreichenbach.

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