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A practical first-order diagnostic framework for assessing wastewater bio-pond performance in infrastructure-limited rural systems

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ABSTRACT

Waste stabilisation ponds (WSPs), also referred to as wastewater bio-ponds, are widely used for low-cost wastewater treatment, but their performance is often difficult to assess due to the coupled influence of bio-logical activity, water quality, pond geometry, hydraulic behaviour, and site-specific operating conditions. This study proposes a practical first-order diagnostic framework for supporting the assessment of WSP systems, particularly where conventional monitoring infrastructure is limited. The framework combines a systems-level understanding of biological, chemical, and physical pond processes with targeted water quality monitoring, field profile measurements, and image-based surface-flow characterisation. Particular attention is given to flow behaviour, since it influences hydraulic residence time, spatial variability, and treatment efficiency. A homography-based image processing workflow is introduced to transform recorded surface motion into a metric reference frame, allowing surface velocity vectors to be estimated from tracked flow features and used as indicative hydraulic information. Because no independent reference-flow measurements was available, the image-derived velocities and discharge estimates are presented as preliminary and indicative first-order hydraulic information for interpreting flow behaviour and approximate discharge conditions, rather than as standalone or quantitatively validated flow measurements. The framework was demonstrated using one surface-flow case study and a single water-quality monitoring campaign at the Woodlands Hills three-pond WSP system, together with a second surface-flow case study at an agricultural runoff channel connected to the Moses River. The results provide time-specific diagnostic insight into hydraulic behaviour and treatment response, but do not constitute replicated, seasonal, or generally representative evidence of WSP performance. The proposed framework therefore provides a practical basis for assessing existing WSP systems, identifying likely performance limitations, and guiding future validation and optimisation work.

Keywords: waste stabilisation ponds, bio-physical systems, bio-pond characterisation, image-based velocimetry, wastewater treatment performance assessment.

INTRODUCTION

Waste stabilisation ponds (WSPs), also referred to as wastewater bio-ponds, are widely used natural wastewater treatment systems that rely on the interaction of physical, chemical, and biological processes to remove organic matter and nutrients from wastewater (Desye et al., 2022; Mburu et al., 2013; von Sperling, 2007). These systems are particularly attractive in developing and rural regions due to their low operational costs, robustness, and ability to function without complex mechanical infrastructure (de Souza and Jack, 2010).

Fundamentally, WSPs operate through the synergistic activity of microbial communities, including bacteria and microalgae, together with hydraulic transport processes, which facilitate the degradation of organic pollutants and the transformation of nutrients (von Sperling, 2007). The performance of these systems, therefore, depends on the interaction between biological processes, pond geometry, environmental conditions, and hydraulic behaviour.

This study proposes a practical first-order diagnostic framework for the characterisation and assessment of wastewater bio-pond systems, with particular relevance to field settings where conventional monitoring infrastructure is limited. The objective is not to present a fully validated standalone performance assessment method, but to provide a structured basis from which system behaviour can be interpreted, likely limitations can be identified, and future optimisation strategies may be informed. Given the limited experimental evidence available, the study is framed as a time-specific field demonstration of the diagnostic framework rather than a replicated or seasonal evaluation of WSP performance. Recognising that performance assessment is most meaningful when interpreted in the context of system characterisation, this work adopts a diagnostic approach in which understanding system behaviour precedes assessment of system performance. To support this, the following section develops a systems-level understanding of waste stabilisation ponds as complex bio-physical systems, outlining the biological, chemical, and hydraulic processes governing their behaviour, common system configurations, and the key parameters relevant to their characterisation and performance assessment.

Within this framework, particular attention is given to hydraulic behaviour and flow dynamics, since these influence residence time, transport processes, and spatial heterogeneity within bio-pond systems. These effects are difficult to quantify directly in situ, as flow conditions are often spatially variable and cannot be represented by a single point measurement. To address this limitation, this study explores an image-based surface-flow estimation workflow for generating indicative surface velocity fields, building on the broader development of video-based and image-based open-channel flow monitoring approaches (Chen et al., 2024; Ljubičić et al., 2024). These velocity fields provide qualitative support for interpreting apparent preferential flow pathways, potential low-flow regions, and spatial variations in surface-flow behaviour; however, such interpretations remain dependent on hydraulic conditions, feature visibility, and surface-feature representativeness (Rozos et al., 2022). The estimated surface velocities may also be combined with approximate cross-sectional information to obtain first-order volumetric flow estimates, which can support residence time calculations where conventional flow measurement infrastructure is unavailable. In this way, the proposed workflow is framed as a practical supporting tool for hydraulic characterisation within the broader performance assessment methodology, rather than as a fully validated standalone flow measurement method.

WASTE STABILISATION PONDS AS COMPLEX BIO-PHYSICAL SYSTEMS

WSPs are inherently complex environmental systems in which biological, chemical, and physical processes interact across multiple spatial and temporal scales (Ho et al., 2018; Liu et al., 2016). Although often described using simplified classifications, their actual behaviour emerges from nonlinear interactions between microbial communities, oxygen dynamics, and hydraulic transport processes. These interactions govern the degradation of organic matter, the transformation of nutrients, and the overall treatment performance of the system.

The primary objective of WSP systems is to improve wastewater quality through the removal of organic matter, nutrients, suspended solids, and pathogens. Organic loading within the system is typically characterised using key indicators such as biochemical oxygen demand (BOD) and chemical oxygen demand (COD), while treatment processes aim to reduce total suspended solids (TSS) and stabilise physicochemical properties of the water (e.g. pH, electrical conductivity). In parallel, nutrient transformation processes focus on the removal or conversion of nitrogen and phosphorus species, which play a critical role in system performance. These parameters form the basis for assessing treatment efficiency and will be examined in greater detail in the following sections (Desye et al., 2022; Mburu et al., 2013; von Sperling, 2007).

As a result, WSPs cannot be fully characterised using isolated parameters alone, but must instead be understood as coupled bio-physical systems in which flow behaviour, environmental conditions, and biological activity are intrinsically linked (Allan and Castillo, 2007; Hauer and Lamberti, 2007; von Sperling, 2007).

Ideal classification of waste stabilisation ponds

WSPs are commonly classified into three primary types: anaerobic, facultative, and aerobic (or maturation) ponds. Anaerobic ponds are characterised by high organic loading rates and operate in the absence of dissolved oxygen, where organic matter is stabilised through anaerobic digestion processes producing gases such as methane (CH_4), carbon dioxide (CO_2), and hydrogen sulphide (H_2S) (von Sperling, 2007). In contrast, aerobic ponds are dominated by oxygen-rich conditions, where organic matter is oxidised by aerobic microorganisms, with oxygen supplied primarily through atmospheric reaeration and photosynthetic activity (de Souza and Jack, 2010). Facultative ponds represent an intermediate pond type in which aerobic and anaerobic processes occur within different zones of the same pond (von Sperling, 2007).

Reality of WSP systems

In practice, however, WSPs rarely behave as completely uniform systems. The classification of a pond as anaerobic, facultative, or aerobic therefore reflects the dominant behaviour of the pond rather than an absolute condition throughout the entire water body. Local variations may still occur due to factors such as depth, mixing, sunlight exposure, sludge accumulation, and hydraulic conditions. As a result, many WSPs develop stratified or spatially variable zones, with facultative ponds representing the most common example of this behaviour (Ho et al., 2018; von Sperling, 2007). As illustrated in Figure 1, this stratification allows different treatment processes to occur simultaneously within the same pond.

Practical configuration of WSP systems

WSP systems are typically configured as a series of treatment units, each designed to fulfil a specific functional role within the overall treatment process. In the most widely adopted configuration, systems begin with anaerobic ponds, which are primarily responsible for the removal of high organic loads through anaerobic digestion processes. These are followed by facultative ponds, where both aerobic and anaerobic mechanisms contribute to further degradation of organic matter. The final stage commonly consists of maturation (or polishing) ponds, which operate under predominantly aerobic conditions and are responsible for pathogen reduction, nutrient polishing, and overall effluent quality improvement (de Souza and Jack, 2010; Peña Varón and Mara, 2004; von Sperling, 2007).

In practice, however, multiple configurations are employed depending on design objectives,

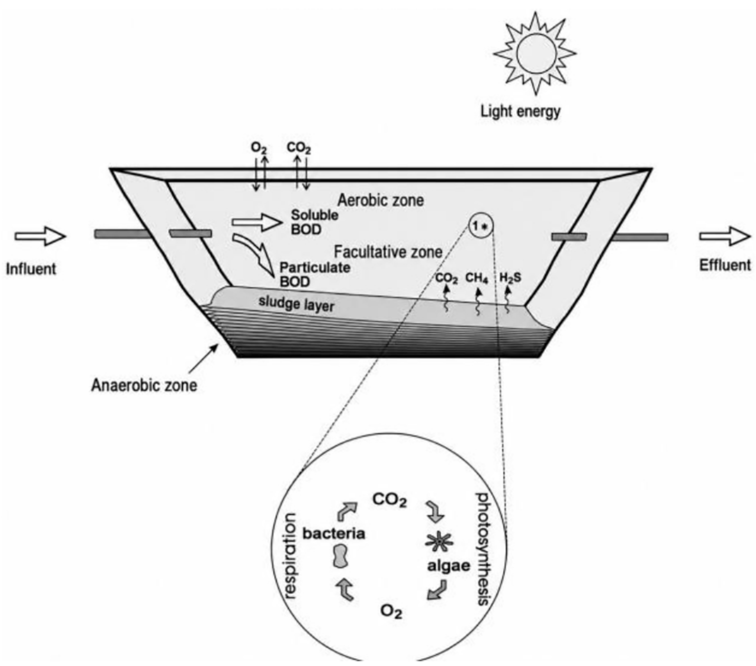


Figure 1. Schematic representation of a facultative waste stabilisation pond illustrating vertical stratification and associated biogeochemical processes. Adapted from (von Sperling, 2007)

organic loading conditions, and site-specific operational constraints (de Souza and Jack, 2010; Peña Varón and Mara, 2004; von Sperling, 2007).

Stratification, oxygen dynamics, and microbial interactions

The behaviour of microbial communities within WSPs is strongly governed by the availability of dissolved oxygen (DO), which regulates whether aerobic or anaerobic degradation pathways dominate. Oxygen is continuously cycled within the system through production by algal photosynthesis, consumption via microbial respiration, and transfer across the air–water interface through reaeration (Ho et al., 2018; Kayombo et al., 2004; Liu et al., 2016).

Importantly, dissolved oxygen should not be interpreted as a direct measure of system performance, but rather as a key indicator of the dominant biological processes occurring within the pond. Variations in DO levels provide insight into prevailing metabolic conditions and are commonly used to distinguish between anaerobic, facultative, and aerobic pond environments. Typical ranges of dissolved oxygen observed in waste stabilisation ponds are summarised in Table 1.

It should be noted that, while strictly anaerobic conditions correspond to zero DO, measured values in real systems are typically non-zero, and concentrations between 0 and 1 mg/L are therefore considered anaerobic-dominant. DO concentrations in surface layers may exceed 8 mg/L under conditions of high algal activity. The ranges presented are indicative and were synthesised from established literature sources (Ho et al., 2018; von Sperling, 2007).

This dynamic oxygen balance is governed by the coupled processes of photosynthesis and respiration, as represented conceptually by the following process expressions:

$$CO_2 + H_2O + \textit{energy} \rightarrow \textit{organic matter} + O_2 \quad (1)$$

$$\textit{organic matter} + O_2 \rightarrow CO_2 + H_2O + \textit{energy} \quad (2)$$

These expressions are not intended as balanced stoichiometric reactions, but rather as simplified process representations adapted from von Sperling (2007). In this context, the organic matter formed in Equation may be interpreted primarily as algal biomass produced through photosynthesis, while the organic matter consumed in Equation 2 represents biodegradable organic material undergoing oxidation within the pond.

These same processes also underpin the microbial interactions responsible for treatment within WSPs. Microalgae produce oxygen through photosynthesis, which supports aerobic bacterial degradation of organic matter, while bacterial respiration releases carbon dioxide that can in turn be utilised by algae. This coupled algae–bacteria interaction forms the biological basis of treatment in facultative and maturation ponds and links oxygen dynamics directly to the breakdown and transformation of wastewater constituents (Kayombo et al., 2004; Sayara et al., 2021).

Table 1. Typical dissolved oxygen (DO) ranges associated with different pond conditions in waste stabilisation systems. Ranges were interpreted and adapted from established literature sources (Ho et al., 2018; Liu et al., 2016; von Sperling, 2007)

Zone / Condition	Typical DO range (mg/L)
Anaerobic (dominant)	0–1
Facultative (bottom layer)	0–1
Facultative (transition layer)	1–2
Facultative (surface layer)	2–8+
Aerobic (well-oxygenated)	≥ 2 (typically > 5 in WSPs)

Organic matter degradation

The degradation of organic matter is one of the primary treatment objectives in WSP systems. In this context, BOD represents the amount of oxygen that would be consumed by microorganisms during the degradation of biodegradable organic matter, while COD represents the amount of oxygen required to chemically oxidise all oxidisable material present in the water, including both biodegradable and non-biodegradable fractions (von Sperling, 2007). Thus, COD may be expressed conceptually as COD = BOD+ non-biodegradable fraction, and therefore COD ≥ BOD (von Sperling, 2007).

Although COD remains useful for characterising the total organic load, the principal focus in WSP systems is typically placed on BOD, since it represents the biologically degradable fraction of the wastewater. Consequently, BOD reduction is a key measure of treatment performance in WSP systems.

A key parameter governing this performance is the BOD loading rate, which represents the mass of biodegradable organic matter applied to the system per unit volume or surface area per unit time. While BOD loading does not directly quantify treatment efficiency, it governs the conditions under which biological treatment processes occur, and therefore strongly influences the overall performance of the pond system.

For anaerobic ponds, the BOD loading is expressed as a volumetric loading rate, while for facultative ponds it is expressed as a surface loading rate. The corresponding relationships are given below (Peña Varón and Mara, 2004):

$$\lambda_v = \frac{L_i Q}{V_a} \tag{3}$$

$$\lambda_s = \frac{10 L_i Q}{A_f} \tag{4}$$

where: λ_v is the volumetric BOD loading rate (g/m^3·d), λ_S is the surface BOD loading rate (kg/ha·d), L_i is the influent BOD concentration (g/m^3), Q is the wastewater flow rate (m^3/d), V_a is the anaerobic pond volume (m^3), and A_f is the facultative pond surface area (m^2).

The permissible ranges of BOD loading for anaerobic and facultative ponds are temperature-dependent and are illustrated in Figure 2, based on Peña Varón and Mara (2004), where the acceptable loading limits are defined as a function of design temperature.

These relationships are fundamental in practice, as they may be used to verify whether an existing system is correctly loaded, to size ponds based on expected influent conditions, and to optimise system operation through adjustment of the flow rate Q. In this way, the BOD loading equations provide a practical link between system design and performance.

Nutrient transformation processes

In addition to organic matter removal, WSPs facilitate the transformation and removal of key nutrients, primarily nitrogen and phosphorus, through a combination of biological, chemical, and physical processes (Kayombo et al., 2004; Verbyla et al., 2017; Zimmo, 2003). The reaction arrows used in this section represent simplified transformation pathways rather than balanced stoichiometric reactions. Nitrogen enters the system predominantly as organic nitrogen, which is converted to ammonium (NH_4^+) through ammonification (Organic N → NH_4^+) (Kayombo et al., 2004; von Sperling, 2007). In aqueous systems, ammonium exists in equilibrium with ammonia (NH_3) according to $NH_3 + H_2O \leftrightarrow NH_4^+ + OH^-$, with the relative proportions dependent on pH and temperature (Emerson et al., 1975). Under aerobic conditions, typically present in the upper water layers, ammonium is oxidised to nitrate via nitrification ($NH_4^+ \rightarrow NO_2^- \rightarrow NO_3^-$) (Kayombo et al., 2004; Zimmo, 2003). In contrast, in the lower anaerobic zones and within the sludge layer, nitrate is reduced through denitrification ($NO_3^- \rightarrow N_2$), resulting in the release of gaseous nitrogen to the atmosphere (Verbyla et al., 2017; Zimmo, 2003).

Phosphorus, primarily present as organic phosphorus in influent wastewater, is mineralised to orthophosphate (PO_4^{3-}) (Organic P → PO_4^{3-}) (von Sperling, 2007). Phosphorus removal occurs

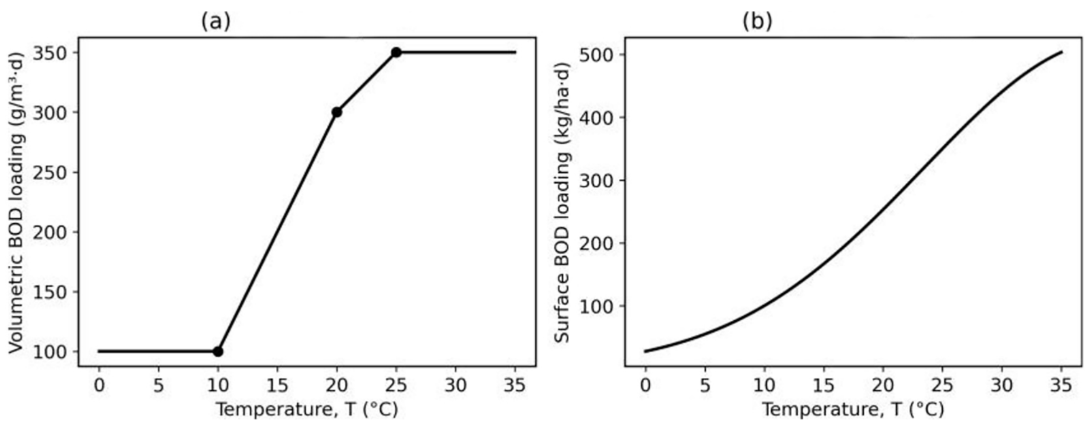


Figure 2. Allowable BOD loading as a function of temperature for waste stabilisation pond design, showing (a) anaerobic pond volumetric loading and (b) facultative pond surface loading, based on Peña Varón and Mara (2004)

through biological uptake by algae and microorganisms, sedimentation associated with biomass and particulate matter, and chemical precipitation with multivalent cations such as calcium, iron, and aluminium, forming insoluble compounds (Kayombo et al., 2004; Verbyla et al., 2017). Full-scale performance studies further show that pond systems can contribute to measurable total phosphorus removal, although the extent of removal depends on system configuration and operating conditions (Desye et al., 2022; Mburu et al., 2013). However, under anaerobic conditions, particularly within the sediment layer, phosphorus may be released back into the water column, reducing overall removal efficiency (de Souza and Jack, 2010; Kayombo et al., 2004).

Common physico-chemical indicators

In addition to nutrient species, routine physico-chemical parameters such as pH, electrical conductivity (EC), and TSS are commonly measured in WSPs, providing practical indicators of the biological, chemical, and hydraulic processes governing system behaviour and treatment performance (Desye et al., 2022; Mara and Pearson, 1987; von Sperling, 2007). Although these parameters are not all direct measures of treatment efficiency, they provide useful supporting information for interpreting algal activity, water chemistry, suspended biomass, settling behaviour, and overall operating conditions (Desye et al., 2022; Liu et al., 2016; Mburu et al., 2013). A summary of these commonly used indicators and their interpretation within WSP systems is presented in Table 2.

Hydraulic behaviour, residence time, and pond geometry

The hydraulic behaviour of WSPs plays a fundamental role in governing treatment performance, as it directly influences residence time, mixing patterns, and the spatial distribution of oxygen and nutrients within the system (de Souza and Jack, 2010; Gopolang and Letshwenyo, 2018; Moreno, 1990). Effective operation of WSPs relies on providing sufficient hydraulic residence time for biological, chemical, and physical processes to occur, including organic matter degradation and nutrient transformation.

Hydraulic residence time is commonly expressed as:

$$\theta = \frac{V}{Q} \tag{5}$$

$$\theta = \frac{A \cdot D}{Q} \tag{6}$$

where: θ is the mean hydraulic residence time (days), V is the pond volume (m^3), Q is the influent flow rate (m^3/day), A is the surface area, and D is the pond depth (de Souza and Jack, 2010; Peña Varón and Mara, 2004).

Equation 5b applies to ponds with approximately rectangular geometry, where pond volume is approximated as $V = A \cdot D$. This relationship highlights the strong dependence of treatment behaviour on both pond geometry and flow conditions.

The use of hydraulic residence time as defined by Equations 5 and 6 provides a useful first-order estimate of system behaviour; however, it implicitly assumes ideal hydraulic conditions, including uniform flow and complete mixing throughout the pond. In reality, WSPs exhibit complex flow patterns characterised by hydraulic heterogeneity, including short-circuiting pathways (highways) and stagnation zones (dead zones), which result in a distribution of residence times rather than a single representative value (de Souza and Jack, 2010;

Table 2. Summary of common physico-chemical indicators in waste stabilisation ponds

Parameter	Represents	Interpretation in WSP systems
pH	Acidity/alkalinity; CO_2 balance	Indicates balance between photosynthesis and respiration; typically increases along treatment due to algal CO_2 uptake; influences nutrient speciation and pathogen removal (Liu et al., 2016; Mara & Pearson, 1987; von Sperling, 2007)
EC	Dissolved ionic content	Indicates the amount of dissolved substances in the water; provides a general measure of water chemistry and system conditions, but is not a direct indicator of treatment performance (Desye et al., 2022; Mara & Pearson, 1987)
TSS	Suspended particulate matter	Indicates presence of solids, biomass, and algae; reduction linked to sedimentation and hydraulic performance; elevated values may indicate poor settling or high algal concentrations (Desye et al., 2022; Mburu et al., 2013; von Sperling, 2007)

Table 3. Typical depth and hydraulic retention time ranges reported for major waste stabilisation pond

Pond type	Typical depth	Typical retention time
Anaerobic pond – typically dark in colour	2–4 m	3–5 days at temperatures above 20 °C; minimum design retention time of about 1 day also reported
Facultative pond – green in colour due to algal growth	1–2 m to about 1.5–3.0 m	2–4 weeks; minimum design retention time of about 4 days also reported
Maturation pond – generally clearer in appearance	1.0–1.5 m, with around 1.0 m often reported as optimal; modern guidance commonly recommends shallow units, generally ≤ 1.5 m	About 12 days; minimum design retention time of about 3 days also reported

Gopolang and Letshwenyo, 2018; Moreno, 1990). Consequently, the actual fluid age within the pond may deviate significantly from the theoretical residence time. Since treatment processes are strongly dependent on the time available for biological and chemical reactions, this discrepancy can lead to reduced treatment efficiency even when nominal design criteria are satisfied. This highlights the importance of considering hydraulic behaviour beyond simplified residence time estimates and motivates the need for more detailed analysis of flow dynamics within WSP systems.

Summary: WSPs as complex coupled bio-physical systems

This section has established a foundational understanding of waste stabilisation ponds as complex, spatially variable systems in which biological, chemical, and physical processes interact to influence treatment behaviour. The classification of ponds into anaerobic, facultative, and maturation units provides a practical framework for interpreting system function, while observable characteristics, such as colour and dissolved oxygen profiles, enable field-based characterisation of dominant pond conditions.

Building on this understanding, key performance indicators have been identified, including BOD reduction and nutrient transformation processes, particularly those governing nitrogen and phosphorus behaviour. Together, these parameters provide a basis for assessing whether a system is operating within expected conditions, when interpreted together with pond configuration, environmental conditions, and hydraulic behaviour.

Importantly, interactions between hydraulic behaviour and biogeochemical processes introduce significant spatial and temporal variability within WSPs, meaning that observed performance may deviate from that implied by simplified design parameters or bulk measurements alone. Metrics such as nominal residence time, loading relationships, and system-averaged water quality indicators provide useful first-order insight, but may not capture the effects of hydraulic heterogeneity, including short-circuiting and stagnation zones. Therefore, practical characterisation requires consideration of both process dynamics and flow conditions, particularly where spatial variability governs treatment effectiveness and influences opportunities for system improvement.

Together, these insights provide the necessary framework for analysing, characterising, and supporting the assessment of existing WSP systems under field conditions.

PROPOSED METHOD

This section presents the proposed diagnostic methodology used to characterise and support the assessment of wastewater bio-pond performance. The method is formulated based on the systems-level understanding developed in the preceding sections, incorporating the coupled biological, chemical, and hydraulic processes influencing waste stabilisation pond behaviour. It integrates two complementary components: image-based surface-flow characterisation and first-order volumetric flow estimation for hydraulic assessment, together with water quality monitoring for system characterisation and performance assessment. Together, these components form a practical field-based framework for producing time-specific diagnostic insight into bio-pond systems.

Surface flow characterisation and volumetric flow estimation

To estimate surface flow from recorded image data, the first step is to convert the camera view into a geometrically consistent metric reference frame. Raw image coordinates do not directly correspond to real-world distances; therefore, a homography transformation is applied to map the image plane onto a metric plane corresponding to the water surface. This is achieved using reference markers with known spatial positions, which are placed near the water surface to support spatial calibration. By doing so, the image is warped such that a fixed spatial resolution is obtained (e.g. 1 pixel = 1 mm) on the measurement plane. Once transformed, the image data can be analysed within a consistent coordinate system.

Following spatial transformation, feature tracking is performed to estimate displacement over time. This can be achieved using a range of image-processing approaches depending on the imaging conditions and available contrast. In the present framework, observable flow features may be isolated using colour-based filtering in the hue–saturation–value (HSV) colour space, greyscale intensity thresholding, or thermal contrast in infrared imagery. The choice of tracking method is therefore application-specific and depends on the available visual or thermal contrast. This is consistent with video-based discharge measurement approaches, where surface ripples, floating objects, or artificial tracers may be used as observable flow features for estimating surface velocity (Chen et al., 2024).

Depending on the tracking approach, observable flow features may be represented by centroids, feature points, or other identifiable image-based markers. In the present study, detected features were represented by their centroids and matched between consecutive frames to estimate displacement vectors. These vectors were then converted to surface velocity estimates using the known temporal resolution of the image sequence. Repeating this process across multiple features and frames yields a set of velocity vectors, forming an indicative surface flow field. The underlying workflow of the proposed method is summarised conceptually in Figure 3.

Surface velocity estimates and the corresponding velocity vector field may be used to support qualitative interpretation of surface-flow patterns within spatially heterogeneous systems and, with approximate flow-depth or cross-sectional information, first-order indicative discharge estimation in situ. However, because no independent reference-flow measurements were available, these outputs are interpreted as preliminary and indicative rather than quantitatively validated, consistent with image-velocimetry studies showing sensitivity to hydraulic conditions, feature visibility, image rectification, and surface-feature representativeness (Rozos et al., 2022).

Characterisation of pond behaviour and water quality-based performance assessment

Water quality measurements in WSP systems provide a diagnostic basis for two complementary objectives:
a) characterisation of pond behaviour, and
b) assessment of treatment performance.

These objectives are inherently interdependent, since observed performance can only be meaningfully interpreted relative to the expected function, internal behaviour, and field conditions of each pond or zone within the system.

Conventional monitoring approaches typically rely on periodic sampling and laboratory analysis of bulk parameters such as BOD, nutrient species, and physico-chemical indicators (de Souza and Jack, 2010; Edok-payi et al., 2021; Kilian et al., 2024; von Sperling, 2007). While such methods provide an overall indication of treatment response, they are often limited in their ability to resolve spatial and temporal variability within the

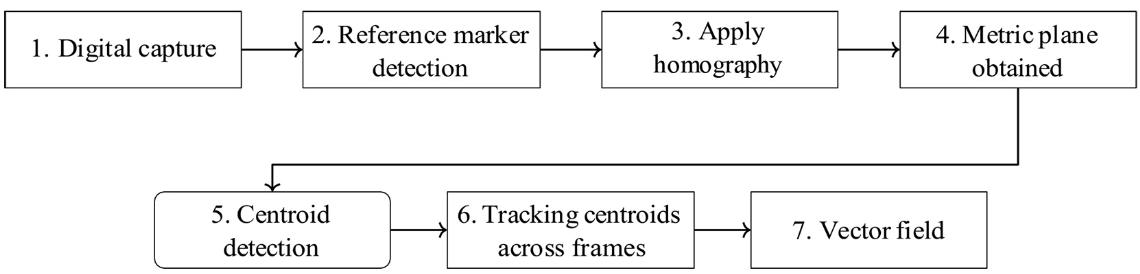


Figure 3. Conceptual workflow of the proposed method for estimating surface flow behaviour and generating a velocity vector field from image data

system. Furthermore, practical constraints related to cost, time, and accessibility restrict the feasibility of high-resolution sampling across large pond systems.

In response, a hierarchical monitoring approach is adopted, in which sampling resolution is adapted to the objectives of the investigation and available resources. This approach begins with low-resolution system-level measurements and may be refined through targeted sampling where required.

Characterisation of pond behaviour

Initial characterisation of individual ponds or zones is performed using a combination of system configuration, geometric characteristics, and in situ measurements. DO profiles provide the primary basis for classification, with typical ranges and stratification behaviour given in Table 1. Supporting information regarding pond geometry, depth, and typical visual characteristics may be obtained from Table 3.

In practice, classification is established by comparing measured DO profiles (e.g., bottom, mid-depth, and surface) with expected stratification behaviour, and supported by observed physical characteristics and system configuration. Additional physico-chemical and nutrient indicators may be used to reinforce classification, as summarised in Table 4.

Performance assessment

Following characterisation, observed system performance is assessed using water quality measurements at the inlet and outlet of each pond. For a system comprising n ponds in series, a practical baseline may be established using $n + 1$ sampling locations, corresponding to the influent of the first pond and the effluent of each subsequent pond.

Performance may be assessed using concentration-based removal efficiency, expressed as the percentage reduction between inlet and outlet concentrations across each pond or the overall system (Desye et al., 2022; Edokpayi et al., 2021; Kilian et al., 2024). In this study, observed performance is primarily assessed through BOD reduction and the transformation and removal of nitrogen and phosphorus species.

Supporting indicators, including TSS, pH, and EC, may provide additional insight into system behaviour, but are not interpreted as standalone measures of treatment efficiency. TSS is particularly useful for assessing sedimentation behaviour and algal presence, while pH and EC assist in interpreting biological activity and overall system conditions (Desye et al., 2022; Mburu et al., 2013; von Sperling, 2007).

In addition to observed performance, the operating condition of each pond may be assessed using established loading relationships. For anaerobic and facultative ponds, the volumetric and surface BOD loading rates may be estimated using Equation 3 and Equation 4, providing a basis for assessing whether the system is operating within expected design conditions (Peña Varón and Mara, 2004).

Targeted spatial monitoring

To gain further insight into internal pond behaviour, additional measurements may be taken within individual ponds at selected regions of interest.

Table 4. Practical indicators for classification of pond behaviour in WSP systems

Indicator	Observed condition	Interpretation
DO (depth profile)	≈0 mg/L throughout the water column	Anaerobic conditions (anaerobic pond or zone)
DO (depth profile)	Low DO at the bottom with increasing DO towards the surface	Stratified conditions (facultative pond behaviour)
DO (depth profile)	Sustained DO in the upper water column	Aerobic conditions (maturation or polishing behaviour)
Nitrate (NO_3^-)	Elevated concentrations	Nitrification occurring in aerobic regions (typically surface layers of facultative or maturation ponds)
Ammonium (NH_4^+)	Persistent concentrations	Reduced conditions with limited nitrification (anaerobic or lower facultative zones)
Nitrogen profile	Decreasing NO_3^- with depth	Denitrification occurring in anaerobic regions
pH	Increasing trend along pond series	Algal activity and CO_2 uptake
TSS	Elevated concentrations	High biomass concentration or poor settling conditions

Due to practical and economic constraints, high-resolution sampling throughout an entire pond is generally not feasible. Instead, where resources allow, hydraulic behaviour may first be screened or independently assessed, for example through Computational Fluid Dynamics (CFD), tracer testing, or flow analysis, before targeted sampling is directed to regions where suboptimal performance is suspected.

This hierarchical monitoring strategy provides a practical balance between diagnostic resolution and sampling feasibility. By combining system-level assessment with targeted spatial measurements informed by hydraulic behaviour, the approach improves representation of internal pond processes, captures spatial variability that may be missed by bulk sampling alone, and supports more informed assessment of bio-pond performance.

RESULTS AND DISCUSSION

To demonstrate the application of the proposed diagnostic framework, the methodology was implemented on a representative waste stabilisation pond system at Woodlands Hills Wildlife Estate, Bloemfontein, South Africa, where hydraulic behaviour and water quality-based performance were assessed under the field conditions present during the measurement campaign. In addition, the image-based surface-flow characterisation component was applied independently at an agricultural runoff channel connecting to the Moses River in Mpumalanga, South Africa, providing a second demonstration site with a higher observed surface velocity. While the proposed framework is formulated in a general and adaptive manner, the available evidence is limited to two surface-flow demonstrations and one WSP monitoring campaign. The following results are therefore presented as time-specific field demonstrations of framework application rather than as replicated evidence of general or seasonal system performance, recognising that WSP behaviour may vary with monitoring period and operating conditions (Kilian et al., 2024).

Surface flow characterisation results

The proposed method was implemented using a FLIR E60 thermal imaging camera, which provides both digital and thermal image data at a resolution of 320 × 240 pixels, as shown in Figure 4(a). Reference markers (ArUco markers in this study) with known spatial positions were used to define the calibration plane, as illustrated in Figure 4(b), enabling the application of a homography transformation based on the digital image. The resulting transformation (Figure 4(c)) was used to obtain a geometrically consistent metric representation of the water surface and was subsequently applied to the corresponding thermal imagery. It should be noted that the marker support structure in Figure 4(a) and (b) differs slightly; however, the configuration remains consistent, with modifications introduced to position the markers closer to the water surface and minimise height discrepancies.

Following spatial calibration, thermal imagery was used to identify and track surface flow features, as shown in Figure 4(d). Centroids corresponding to detectable thermal contrast regions were extracted and tracked across consecutive frames. The effective frame rate was determined directly from the recorded video using custom Python processing to reduce reliance on the nominal frame rate. Displacement between frames, combined with this measured frame rate, was used to estimate surface velocity vectors, resulting in a

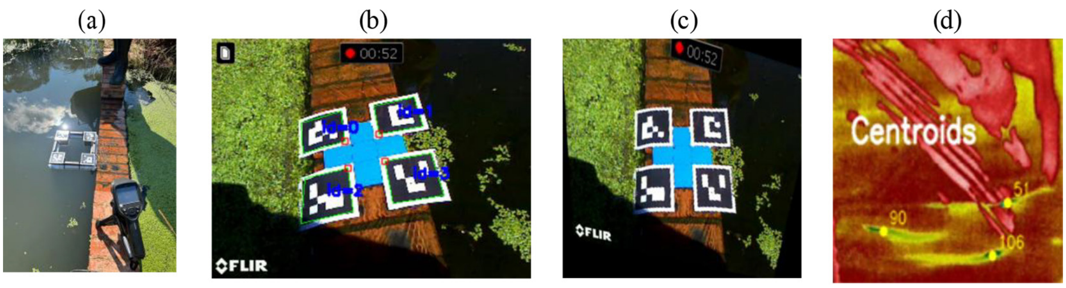


Figure 4. Implementation of the proposed flow estimate method: (a) experimental setup and measurement channel, (b) detection of reference markers used for spatial calibration, (c) application of homography to obtain a metric representation, and (d) detection of centroids from thermal imagery for tracking

spatially distributed indicative surface flow field. To support flow estimation consistency, the camera was maintained in a fixed position throughout all tests, as shown in Figure 4(a). The resulting flow fields are presented in the following section.

Case Study 1: Woodlands hills bio-pond outlet

The image-based workflow was applied at the outlet of a constructed bio-pond at Woodlands Hills Wildlife Estate. The resulting indicative velocity vector field is shown in Figure 5. The estimated surface flow exhibited low velocity magnitudes, with an estimated mean surface velocity of approximately 0.2781 m/s.

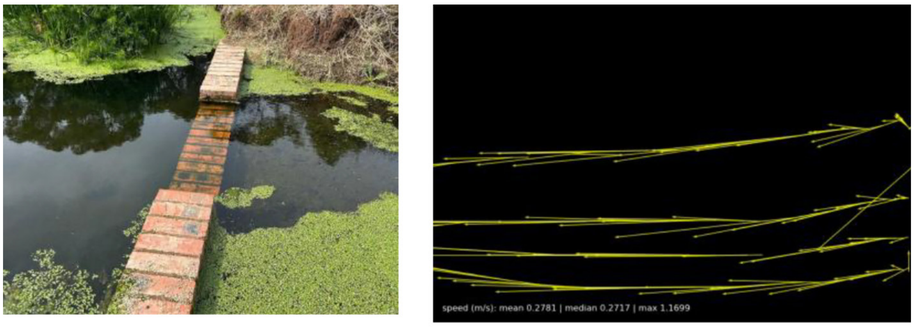
Case Study 2: Agricultural runoff channel to the Moses River

The image-based workflow was subsequently applied at an agricultural runoff channel connected to the Moses River. The resulting indicative velocity vector field is shown in Figure 6. A higher surface velocity magnitude was observed than in Case Study 1, with an estimated mean surface velocity of approximately 0.8676 m/s.

Pond characterisation and performance assessment results

To characterise pond behaviour and evaluate treatment performance, spatially distributed field profile measurements together with laboratory water quality analyses were conducted across the Woodlands Hills bio-pond system. These field and laboratory measurements were collected during a single monitoring campaign; temporal replicate sampling campaigns and seasonal monitoring were not conducted. Sampling locations and field profile measurement points are shown in Figure 7. Based on the general configuration principles of waste stabilisation pond systems discussed previously, the Woodlands Hills bio-pond system would be expected to operate as a series comprising an initial anaerobic pond, followed by a facultative pond, and a final maturation pond, providing a baseline for interpreting the observed pond behaviour.

In accordance with the diagnostic framework proposed in the previous section,



(a) (b)

Figure 5. Case Study 1 measurement site and results, showing (a) the outlet measurement location and (b) the corresponding velocity vector field

(a) (b)

Figure 6. Case Study 2 measurement site and results, showing (a) the measurement location and (b) the corresponding velocity vector field

laboratory sampling locations (S1–S4) were selected following an $n + 1$ sampling approach for a system of n ponds in series, with measurements taken at the influent of the first pond and at the outlet of each successive pond. These locations were used primarily for performance assessment through nutrient and water quality analysis.

Field profile measurement points (FP) were used primarily for pond characterisation through depth-resolved DO profiling. Three field profile locations were selected in each pond, including two points positioned in areas visually interpreted as likely preferential-flow regions and one point in an area interpreted as a likely low-flow or stagnation region, to capture spatial variability and stratification behaviour within the system.

Field profile characterisation results

DO depth-profile measurements used for field characterisation are summarised in Table 5. In this study, DO profiles were obtained using a multiparameter water quality probe (Yosemitech Y4001) connected to a handheld meter (Y600-A). The probe was deployed from the pond bank using a fixed support system (fishing-rod based), allowing controlled and incremental lowering into the water column.

Measurements were taken starting at approximately 50 mm below the water surface, with subsequent readings obtained at intervals of approximately 150 mm. At each depth, the probe was held in position until the dissolved oxygen reading stabilised, after which a single stabilised value was recorded. The profiles therefore provide depth-resolved field observations for the monitoring campaign, but do not represent repeated temporal profiles or statistical replicates at each depth. Due to variations in pond depth and field conditions, the number of measurements differs between locations. Dissolved oxygen concentrations are reported in mg/L.

Nutrient and water quality performance assessment

Water-quality samples at S1–S4 were collected as single grab samples during the same monitoring week as the field profile measurements and were transported to the analytical laboratory in a cooler bag immediately after collection. Samples were analysed in accordance with ISO/IEC 17025-accredited laboratory procedures. The samples were not composite or temporally replicated samples; consequently, the laboratory results represent instantaneous concentrations for the monitoring period rather than long-term or seasonal water-quality conditions (Table 6).

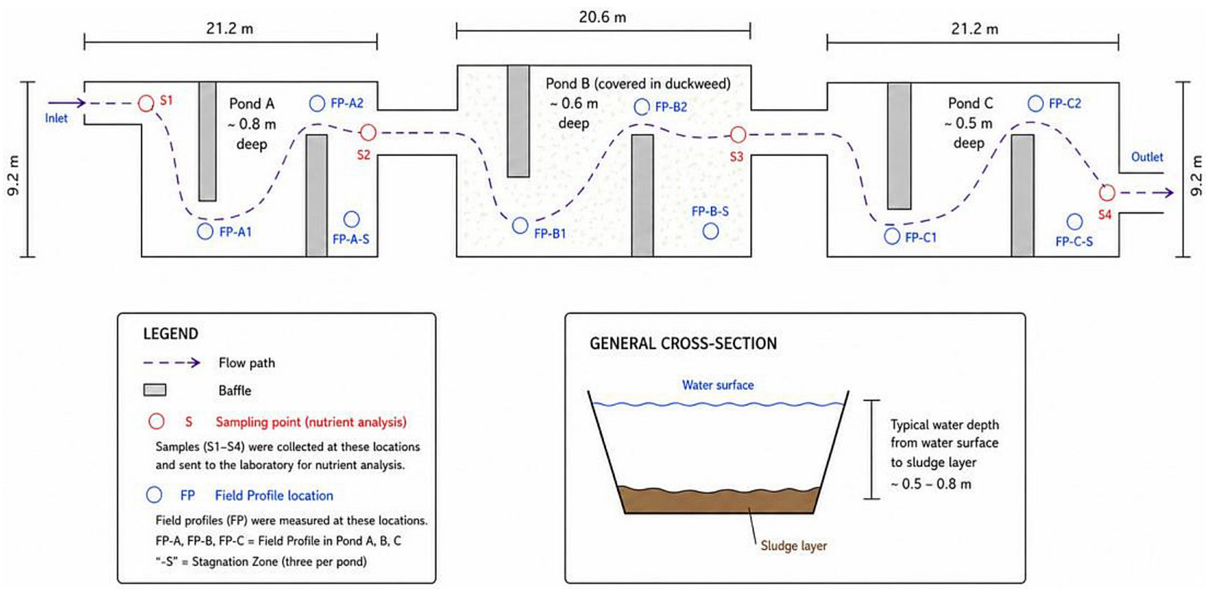


Figure 7. Schematic representation of the Woodlands Hills bio-pond system showing laboratory sampling locations (S1–S4) used for performance assessment and field profile measurement points (FP) used for pond characterisation. Flow-path and low-flow region annotations were interpreted from site geometry and field observations and were used to guide diagnostic sampling. Geometry was based on field measurements and illustrated using an AI-assisted schematic

Table 5. Depth-profile dissolved oxygen measurements at field profile locations shown in Figure 7. Depth values indicate approximate distance below the water surface. Blank entries indicate that the corresponding depth was not reached.

Dissolved oxygen (DO) measurements (mg/L)										
Location	Depth below surface (mm)									Notes
	50	200	350	500	650	800	950	1100	1250	
FP-A1	0.05	0.03	0.03	0.02	0.02	–	–	–	–	Covered in duckweed
FP-A2	0.56	0.56	0.52	0.43	0.36	0.17	0.11	–	–	Open surface
FP-A-S	0.49	0.47	0.42	0.39	0.37	0.33	0.30	0.36	–	Open; stagnation zone
FP-B1	0.55	0.53	0.52	0.51	0.41	0.23	0.24	–	–	Covered in duckweed
FP-B2	0.71	0.64	0.58	0.58	0.54	0.54	0.53	0.50	0.47	Open surface
FP-B-S	0.36	0.27	–	–	–	–	–	–	–	Very shallow (~20 cm), dense vegetation
FP-C1	0.35	0.32	0.32	0.31	0.30	0.27	0.23	–	–	Covered in duckweed
FP-C2	0.41	0.42	0.69	1.39	1.56	1.69	–	–	–	Open; rainfall during measurement
FP-C-S	1.38	1.24	1.21	1.29	1.29	1.56	1.23	–	–	Semi-open; rainfall during measurement

Note: Measurements were conducted at an ambient temperature of approximately 17 °C under overcast and lightly windy conditions. No rainfall occurred during the preceding 72 hours. All profiles were measured under dry conditions, except for FP-C2 and FP-C-S, which were recorded immediately after a brief 1–2 minute rainfall event during the latter part of the campaign.

Surface flow characterisation discussion

The proposed workflow produced physically plausible velocity-vector patterns at both demonstration sites; however, plausibility does not constitute validation against true or reference flow conditions. Because no independent reference-flow measurements were available for this study, the results should be interpreted as preliminary and indicative rather than quantitatively validated. This interpretation is consistent with recent image-velocimetry studies that present image-based methods as promising tools for open-channel surface velocity and flow-rate estimation, while also emphasising sensitivity to field conditions, feature visibility, image rectification, surface-feature representativeness, and the need for validation (Chen et al., 2024; Ljubičić et al., 2024; Rozos et al., 2022). Estimated mean surface velocities of approximately 0.28 m/s and 0.87 m/s were derived from the processed image data and may support first-order discharge approximation when combined with cross-sectional estimates.

The observed vectors were generally aligned with the dominant flow direction, suggesting that the workflow can provide qualitative insight into apparent surface-flow pathways and local variation in surface movement within the measurement sections. This interpretation remains indicative, rather than confirmatory, because individual vector estimates are affected by feature visibility, tracking quality, and local image-processing uncertainty.

Some local vector irregularities or apparent jumps were observed, primarily due to limitations associated with masking and centroid detection during feature tracking. These irregularities do not necessarily prevent qualitative or first-order interpretation, but they limit the precision of individual vectors. Interpreting the vector field at an aggregate level may reduce sensitivity to individual outliers, provided suitable quality-control filters are used.

Volumetric flow estimation discussion

The image-derived surface velocities were used only to obtain first-order indicative discharge approximations using:

$$Q = \bar{v}A \tag{7}$$

where: $\bar{v}$ is the estimated mean surface velocity and A is an approximate flow cross-sectional area.

This follows the general velocity–area principle, but is not presented as an ISO 748-compliant or independently validated discharge measurement (International Organization for Standardization, 2021). The approximation depends on simplified cross-sectional geometry and on the assumption that the tracked surface motion is

Table 6. Laboratory water quality results for sampling locations S1–S4. Values below detection limits are indicated by ”<”. Blank entries indicate parameters that were not analysed at the corresponding sampling location

Parameter	Unit	S1 Pond A inlet	S2 Pond A outlet	S3 Pond B outlet	S4 Pond C outlet
pH	pH units	7.11	7.31	7.21	7.24
Ammonia (NH_3-N)	mg/L	< 1	< 0.1	< 0.1	< 1
Ammonium (NH_4^+-N)	mg/L	6.30	5.40	4.60	4.00
Biochemical oxygen demand (BOD_5)	mg/L	25.00	23.00	19.00	16.00
Chemical oxygen demand (COD)	mg/L	37.00	–	–	33.00
Nitrate (NO_3^--N)	mg/L	< 1	< 1	< 1	< 1
Nitrite (NO_2^--N)	mg/L	0.150	0.080	0.040	0.030
Orthophosphate (PO_4^{3-}-P)	mg/L	12.70	< 3.7	11.30	2.90
Total ammonia nitrogen (TAN)	mg/L	7.0	6.4	5.6	5.0
Total phosphate (TP)	mg P/L	4.10	–	–	0.90
Total nitrogen (TN)	mg/L	8.45	–	–	6.03

sufficiently representative of bulk flow. These assumptions are particularly important in shallow or irregular channels, where small errors in flow thickness can produce large relative changes in estimated discharge.

Estimated discharge ranged from approximately 0.00115–0.00229 m^3/s for the Woodlands Hills bio-pond and 0.068–0.090 m^3/s for the agricultural runoff channel to the Moses River. These values should be interpreted as indicative hydraulic estimates rather than precise flow measurements, and are used here to illustrate the workflow's potential for supporting preliminary in situ flow assessment (Table 7).

Uncertainty, reproducibility, and validation requirements of the proposed method

The proposed method is field-based and therefore subject to uncertainty from both the water quality assessment and the image-based hydraulic workflow. For the water quality and field profile component, uncertainty is mainly associated with probe accuracy, sensor stabilisation, measurement depth, sample handling, and laboratory analytical uncertainty, as reflected in recent WSP monitoring studies using field and laboratory measurements (Edokpayi et al., 2021; Kilian et al., 2024). The DO sensor used for field pro-filing has a manufacturer-stated accuracy of ±1% or ±0.3 mg/L maximum (Yosemite Technologies Co., Ltd., 2022). These factors support interpreting the pond classification and performance assessment as time-specific results obtained during the sampling campaign, rather than as a general long-term assessment of system performance. The experimental evidence is limited to two surface-flow case studies and one WSP monitoring campaign; therefore, the pond classifications, removal efficiencies, and hydraulic interpretations should be treated as instantaneous diagnostic evidence rather than long-term, seasonal, or statistically replicated performance indicators.

For the image-based surface-flow workflow, uncertainty is introduced progressively through the processing chain. Spatial uncertainty arises from marker placement, marker detection, the assumed calibration plane, homography accuracy, lens and perspective effects, and thermal–digital image alignment. Temporal uncertainty is associated with the effective video frame rate and any frame-rate variation. Feature-detection uncertainty depends on surface-feature visibility, thermal or visual contrast, threshold selection, mask cleaning, blob-size filtering, and centroid extraction. Tracking uncertainty is introduced when centroids are matched between consecutive frames, particularly where features merge, split, disappear, or produce apparent jumps. Finally, the conversion from surface velocity to discharge introduces additional uncertainty because it assumes that tracked surface motion is representative of bulk flow and because flow width and mean flow thickness are approximated. These sources are consistent with reported limitations of video-based and image-based open-channel flow measurement workflows, where accuracy depends on hydraulic conditions, image rectification, feature characteristics, and surface-feature

Table 7. First-order indicative discharge estimates for the two flow case studies

Location	Woodlands hills bio-pond	Ag. runoff (Moses R.)
Estimated mean velocity (m/s)	0.28	0.87
Measured flow width (m)	1.03	2.60
Estimated mean flow thickness (m)	0.004–0.008	0.030–0.040
Estimated discharge (m^3/s)	~ 0.00172	~ 0.0790
Estimated discharge (L/s)	~ 1.72	~ 79.0

Note: Flow width was measured in the field using a tape measure. Mean flow thickness was estimated using direct ruler measurements at the measurement section and is reported as an approximate range due to local variation in water depth. Discharge values were calculated from the estimated mean surface velocity and approximate cross-sectional area and should be interpreted as first-order indicative estimates.

representativeness (Chen et al., 2024; Ljubičić et al., 2024; Rozos et al., 2022).

In terms of reproducibility and practical limitations, the water quality and field profile components are relatively reproducible where field probes, suitable sampling bottles, and laboratory analysis are available. The image-based component is more site-dependent. It requires a fixed camera position, sufficient visual or thermal contrast for tracking surface features, and reference points of known geometry positioned close to the water surface for homography calibration. These requirements may be difficult to satisfy in wide wetlands, inaccessible channels, or infrastructure-limited rural sites where stable camera mounting, marker placement, or suitable tracer contrast cannot be achieved. In addition, the workflow requires appropriate image-processing tools and, depending on the implementation, access to digital and thermal imaging equipment. This is consistent with recent image-velocimetry software studies, which emphasise workflow transparency and preprocessing as important for reproducible flow estimates (Ljubičić et al., 2024).

The present study did not include direct validation against an independent reference-flow measurement technique. Consequently, no combined numerical accuracy band is assigned to the image-derived velocities or discharge estimates. Future validation should compare image-derived surface velocities and discharge estimates with reference measurements obtained under the same hydraulic conditions, for example using current-meter or acoustic Doppler velocity measurements, calibrated weirs or flumes, or controlled flume measurements with known discharge (International Organization for Standardization, 2021; Turnipseed and Sauer, 2010). Such validation is required before the image-based workflow can be used as a standalone quantitative flow measurement technique.

Characterisation of pond behaviour

The dissolved oxygen profiles in Table 5 were interpreted using the DO ranges in Table 1 and the practical classification indicators in Table 4. Measurements were conducted under overcast conditions at an ambient temperature of approximately 17 °C, which likely reduced photosynthetic oxygen production. At the time of measurement, Pond B was largely covered by duckweed, while Ponds A and C were partially covered. Therefore, the results represent an instantaneous field characterisation of pond conditions during the sampling campaign, rather than evidence of seasonal behaviour, normal operating conditions, or long-term pond classification.

Pond A exhibited consistently low DO concentrations, with values below 1 mg/L across all locations. The duckweed-covered region (FP-A1) showed near-zero DO throughout, while the open regions (FP-A2 and FP-A-S) showed slightly higher values but still remained within anaerobic ranges. Overall, Pond A was classified as anaerobic under the measured field conditions.

Pond B also showed predominantly anaerobic behaviour, with DO values below 1 mg/L at most locations, including the open surface location (FP-B2). Since Pond B would typically be expected to exhibit facultative behaviour, this indicates suboptimal performance under the measured conditions. The extensive duckweed cover likely limited light penetration and reduced photosynthetic oxygen production.

Pond C displayed the greatest spatial variability. The duckweed-covered region (FP-C1) remained anaerobic, with DO < 0.4 mg/L, while the open and semi-open locations (FP-C2 and FP-C-S) showed higher DO concentrations, with some values exceeding 1 mg/L and indicating a partial shift towards facultative conditions. However, FP-C2 and FP-C-S were measured immediately following a brief rainfall event, which likely disturbed the water column and produced atypical profiles. These results should therefore be interpreted cautiously and not directly compared to undisturbed profiles. Overall, Pond C showed improved oxygenation potential in open regions, but remained underperforming relative to the expected function of a maturation pond.

The dissolved oxygen-based classification is further supported by the nitrogen species reported in Table 6. Since ammonium, nitrite, and nitrate are reported as nitrogen-equivalent concentrations, they may be compared directly. Ammonium remained present throughout the system, decreasing from 6.30 mg/L at S1 to 4.00 mg/L at S4, while nitrate remained below 1 mg/L at all sampling locations. This indicates limited ammonium-to-nitrate conversion and suggests restricted nitrification under the predominantly low-oxygen conditions observed during the measurement campaign.

Overall, the system was largely anaerobic-dominant during the measurement campaign. Duckweed-covered areas consistently exhibited lower DO, while open regions showed greater oxygenation potential. In addition to duckweed cover, hydraulic effects may have contributed to the observed spatial variability, although internal pond hydraulics were not directly quantified. Poor mixing, low-flow regions, or preferential-flow pathways can limit oxygen distribution and reduce effective contact between wastewater and active treatment regions. These observations therefore suggest that duckweed control, together with targeted hydraulic assessment and possible improvement of flow distribution, may improve system oxygenation, particularly in Ponds B and C.

Initial performance assessment

Initial performance assessment of the biopond system was conducted in accordance with the water quality-based performance assessment approach described in the proposed method. Laboratory results obtained at sampling locations S1–S4 were used to assess concentration changes between the inlet of the first pond and the final outlet of the system. This provided a practical first-order assessment of treatment response during the sampling campaign only; the available data do not resolve temporal or seasonal variability in removal

Table 8. Overall system performance based on inlet (S1) and final outlet (S4) concentrations during the sampling campaign

Parameter	Inlet (S1) [mg/L]	Outlet (S4) [mg/L]	Reduction (%)
Biochemical oxygen demand (BOD)	25	16	36.0
Chemical oxygen demand (COD)	37	33	10.8
Total nitrogen (TN)	8.45	6.03	28.6
Total phosphate (TP)	4.10	0.90	78.0

Table 9. Pond-by-pond BOD reduction, estimated BOD loading rates, and approximate allowable loading comparison during the sampling campaign

Pond / Section	Performance			Loading comparison	
	Inlet [mg/L]	Outlet [mg/L]	BOD reduction [%]	Estimated loading	Approx. allowable loading
Pond A (S1–S2)	25	23	8.0	λ_v = 23.8 g/m^3 · d	≈ 250 g/m^3 · d
Pond B (S2–S3)	23	19	17.4	λ_s = 180.4 kg/ha · d	≈ 200 kg/ha · d
Pond C (S3–S4)	19	16	15.8		
Overall system	25	16	36.0		

Note: Loading values are first-order estimates based on the estimated flow conditions used in the analysis. Approximate allowable loading values were estimated from Figure 2 at 17 °C. Pond A uses volumetric loading and Pond B surface loading; no Pond C loading is shown because Equations 3 and 4 do not define a maturation pond BOD loading formulation.

performance before assessing the distribution of treatment across individual ponds (Table 8).

The results indicate moderate removal of biodegradable organic matter during the sampling campaign, with a 36% reduction in BOD across the pond series. In contrast, COD reduction was limited, suggesting only partial removal of total oxidisable material. Nutrient removal exhibited greater variability, with total nitrogen decreasing by approximately 29%, while total phosphate showed substantial removal of 78%.

To further assess the distribution of treatment response within the system, BOD reduction was assessed on a pond-by-pond basis, as shown in Table 9. In addition, BOD loading rates were estimated using Equations 3 and 4 to relate observed performance to the estimated operating loading conditions within each pond.

The pond-by-pond analysis shows that the largest single BOD reduction occurred in Pond B, while Pond A exhibited limited removal, indicating underperformance during the sampling campaign. When considered alongside the estimated loading conditions, this behaviour is consistent with Pond A operating under substantially underloaded conditions, which may limit the efficiency of anaerobic degradation processes. In contrast, Pond B operated within a loading range close to typical design values and corresponded with greater BOD removal than Pond A. These results should therefore be interpreted as a time-specific assessment under the measured field and loading conditions, rather than as a replicated assessment across seasons, loading states, or long-term operating conditions.

CONCLUSIONS

This study presented a practical first-order diagnostic framework for supporting wastewater bio-pond performance assessment by integrating image-based hydraulic characterisation, field-based dissolved oxygen profiling, and targeted water quality analysis within a systems-level interpretation. The framework was illustrated through a time-specific field application rather than a replicated or long-term performance evaluation. Its value therefore lies in combining multiple indicators to interpret pond behaviour under the measured campaign conditions, rather than in any single measurement or standalone diagnostic output.

Application of the framework to the Woodlands Hills bio-pond system indicated that the ponds were largely anaerobic-dominant during the measurement campaign, with limited nitrification and reduced oxygenation in duckweed-covered regions. This interpretation was supported by persistent ammonium, low nitrate concentrations, and dissolved oxygen profiles measured under overcast field conditions. The pond-by-pond – BOD assessment further showed a time-specific, non-uniform treatment response, with limited removal in Pond A and greater removal in Pond B, consistent with the estimated loading conditions. These findings suggest that duckweed control or removal may be a practical first intervention to improve light penetration and oxygenation, particularly in Ponds B and C, although the response should be confirmed through further monitoring.

The image-based surface-flow workflow provided preliminary hydraulic insight at the two demonstration sites, but the velocity fields and discharge estimates remain indicative because no independent reference-flow measurements were available. The workflow may support screening of apparent surface-flow pathways and approximate discharge conditions where conventional flow measurement infrastructure is limited. However, standalone quantitative use requires validation against reference measurements under comparable hydraulic conditions and improved characterisation of surface-to-bulk velocity and cross-sectional assumptions.

Overall, the study illustrates a practical, time-specific diagnostic pipeline for linking flow behaviour, pond classification, loading conditions, and water quality response in infrastructure-limited WSP settings. The framework can help identify likely performance limitations and guide targeted validation or optimisation, but should not be interpreted as a general, seasonal, or fully validated assessment of long-term pond performance.

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