### Applications of image processing for performance assessment of wastewater treatment plants

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Applications of Image Processing for Performance Assessment of Wastewater Treatment Plants

A dissertation presented
by
Darragh Mullins
to
The College of Engineering and Informatics
in fulfillment of the requirements
for the degree of
Doctor of Philosophy
in the subject of
Electrical and Electronic Engineering

National University of Ireland Galway
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September 2018
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iv
Applications of Image Processing for Performance Assessment of Wastewater Treatment Plants

Abstract

This thesis is concerned with the development of methodologies for automatically assessing fluid properties in wastewater treatment plants, and to monitor plant performance and efficiency. Reliable performance metrics are important for optimising the operation of treatment plants. While sensor technologies are not new to the wastewater treatment field, the application of imaging and image processing systems has seen little development in recent years. Currently, there are a multitude of sensor technologies to monitor a variety of performance parameters. However, research has found that, in practice, the results from these sensors are rarely relied upon due to inconsistent calibration schedules, compounded by a lack of full-time maintenance staff. Instead, these sensors are frequently bypassed in favour of more subjective manual estimates.

In this thesis, the problems with current monitoring practices for wastewater treatment plants are considered. Two areas of interest are highlighted; (i) sludge monitoring, using Sludge Volume Index as a performance metric and (ii) effluent monitoring, using turbidity as a quality metric. A review of the guideline sludge monitoring procedures is presented as well as a discussion of alternative on-site monitoring practices. This review highlights the need for an automated system for performing sludge monitoring, as per the guideline procedures. Subsequently, an image processing system for settled sludge volume measurement is proposed and tested.

Effluent turbidity is primarily affected by a high number of colloidal particles in the particle size distribution. Current effluent turbidity monitoring practices include submerged turbidimeters, that require regular calibration. In practice, it was found that subjective manual estimation of turbidity was often being conducted "by eye". A study was devised to characterise the accuracy of the subjective estimation.
Firstly, an imaging methodology was designed to capture effluent images illustrating light decay as a function of increasing fluid depth. These images were then presented to persons with no knowledge of wastewater treatment monitor and subjectively rated on turbidity. The results of the subjective test were then compared to established laboratory-based turbidity measurement, and a clear correlation was found between the two. Subsequently, an image processing system was designed to replace the observer and to objectively characterise the light decay as a function of fluid depth. Once again, the results from this image processing system were compared to the established laboratory-based measurements and an improved correlation over the subjective comparison was found. Finally, the implications of the deployment of a combined monitoring system is discussed, along with the benefits to current monitoring practices.
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## Glossary of Terms

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<td>APHA</td>
<td>American Public Health Association</td>
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<tr>
<td>AutoSSV</td>
<td>SSV measurement determined by image processing</td>
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<td>BOD</td>
<td>Biochemical Oxygen Demand</td>
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<tr>
<td>CCC</td>
<td>Lin’s Concordance Correlation Coefficient</td>
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<td>CET</td>
<td>Camera Estimated Turbidity</td>
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<tr>
<td>COD</td>
<td>Chemical Oxygen Demand</td>
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<tr>
<td>DO</td>
<td>Dissolved Oxygen</td>
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<td>EPA</td>
<td>Environmental Protection Agency</td>
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<tr>
<td>FAU</td>
<td>Formazin Attenuation Units</td>
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<tr>
<td>IR</td>
<td>Infrared</td>
</tr>
<tr>
<td>MLSS</td>
<td>Mixed Liquor Suspended Solids</td>
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<tr>
<td>NTU</td>
<td>Nephelometric Turbidity Unit</td>
</tr>
<tr>
<td>ORP</td>
<td>Oxidation Reduction Potential</td>
</tr>
<tr>
<td>PE</td>
<td>Population Equivalent</td>
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<tr>
<td>PP</td>
<td>Particle Properties</td>
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<td>PSD</td>
<td>Particle Size Distribution</td>
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<td>SS</td>
<td>Suspended Solids</td>
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<td>SSV</td>
<td>Settled Sludge Volume</td>
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<td>APHA Standard Methods for the Examination of Water and Wastewater</td>
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<td>TIFF</td>
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<tr>
<td>TSS</td>
<td>Total Suspended Solids</td>
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<td>UV</td>
<td>Ultraviolet</td>
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<td>WFD</td>
<td>Water Framework Directive</td>
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<td>WWTP</td>
<td>Wastewater Treatment Plant</td>
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Statement of Originality

I hereby declare that the work contained in this thesis has not been submitted by me in the pursuance of any other degree.

Name: [Signature]

Date: 05/02/2019
Sponsor Acknowledgement

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Chapter 1

Introduction

1.1 Motivation

Wastewater quality monitoring and assessment is a key area for future development due to the high energy costs associated with treatment and management of water resources both nationally and internationally [1, 2]. The management of global water resources has social, policy, and public health implications. In Ireland, wastewater treatment plants (WWTPs) of a size greater than 500 P.E. (Population Equivalent) must be licensed by the Environmental Protection Agency (EPA); the discharge limits for the various parameters of the treated effluent are detailed in the licence and non-conformance with these limits results in significant fines.

Stricter discharge regulations imposed by European regulatory bodies will present challenges in the coming years, not only to conform with these lower discharge lim-

1
its, but also to reduce the associated operating costs by improving operational efficiency. Development and implementation of cost-effective approaches to performance monitoring, for physical, chemical, biological, and ecological processes, that can be standardised across all water treatment facilities could be key to improving resource management, particularly for small to medium-scale plants that do not have full-time maintenance staff on site [3, 4].

A reduction in discharge limits from wastewater treatment plants equates to an increase in the treatment time required and as a result, increased energy consumption. More emphasis on plant operation and monitoring procedures is required to reduce energy consumption. In 2012, the EPA reported that in Ireland, 94% of national waste water received at least secondary treatment, while only 57% of the country’s 541 identified licensed urban WWTPs achieved all of the required effluent quality and sampling standards. These reports indicate the difficulties that exist in achieving discharge limits for which there is a legislative imperative to attain full compliance. Facilities can struggle to comply with regulations and can be inefficient due to infrequent monitoring and the lack of remote monitoring systems [5, 6]. In the absence any such legislative requirements, there is little or no evidence of any regular monitoring or test programmes in place at WWTPs to optimise the efficiency.

The importance of developing performance indicators to ensure proper monitoring is recognised internationally, in particular the delivery of real-time infrastructural services (International Organisation for Standardisation (ISO), 24511:2007). To this
end, the development of a novel approach to low cost monitoring that could be easily deployed in WWTPs is required. Incorporating low cost and robust monitoring systems into current WWTP maintenance schedules is key to improved facility efficiency, greater legislative compliance and potentially reduced energy consumption.

The Water Framework Directive (WFD) [7] along with the INSPIRE Directive (Infrastructure for spatial information in Europe) [8] highlight the importance of efficient data management and effective control and monitoring for reliable WWTP performance. Effective control can prove difficult for remotely located treatment plants, where low-bandwidth (or no connectivity) data connections inhibit the ability to reliably deploy a full real-time monitoring solution. In these situations, not only is complying with regulatory standards challenging, but operating the plant efficiently on a day to day basis is also difficult due to the lack of ongoing plant performance information.

There is a high cost associated with travel of operators to remotely located plants. Development of low-cost, robust, and data-efficient remote monitoring systems would allow operators to make more informed decisions regarding plant operation, thus avoiding unnecessary journeys to plants that are operating within acceptable discharge limits. Equally, improved monitoring could forestall a problem that might otherwise arise between routine maintenance visits.

For activated sludge-based waste water treatment, it is known that the most important operational difficulty is the treatment and reduction of sludge; an operation
that is generally monitored by sludge settlement testing [9]. The settleability of sludge is greatly affected by the size and shape of the particulate matter present in the sludge. Examining the Particle Size Distribution (PSD) could yield interesting results about sludge composition and treatability.

One example of a monitoring technique, examined in this thesis, is the use of turbidity of the effluent stream as a performance indicator for water and wastewater treatment facilities [10, 11] as well as a surrogate for other performance metrics [12, 13]. Effluent quality measurement is a key factor in WWTP operations; (i) for assessing operational efficiency and (ii) for ensuring regulatory compliance. However, in smaller less centralised treatment plants it is often the case that the only monitoring conducted is the minimum required by regulation [14, 15]. The frequency of this monitoring is less than monthly in some cases, often leading to non-compliance with discharge limits [16, 17]. Therefore, there is potential for poor operational efficiency due to lack of real-time performance indicators [16, 18]; the lack of permanent on-site staff compounds the problem.

Specific sensor systems have been developed to measure key wastewater effluent parameters such as total suspended solids (TSS), chemical oxygen demand (COD), biochemical oxygen demand (BOD), nitrate and phosphorous levels. However, these sensors require regular calibration to be reliable [19–21]. Turbidity has proven to be a good surrogate for the same key parameters [22–25], therefore an effluent turbidity monitoring system would be a useful tool for taking surrogate measurements of these
parameters.

1.2 Anatomy of a Wastewater Treatment Plant

WWTPs employ a series of processes to treat or remove harmful elements from wastewater (such as nutrients, that stimulate the growth of aquatic plants, or other toxic compounds and pathogens) to a safe level for discharge to the surrounding environment [26]. Evaluation of these treatment processes is key to optimising plant performance.

There are a number of phases to the treatment process; the most common treatment system present in Ireland is the Activated Sludge Process [27]. This can be described by a number of key elements (as shown in Figure 1.1):

- Preliminary Treatment: in most cases, this includes mechanical screening and grit separation to remove objects and insoluble particles which may damage downstream equipment and processes. Figure 1.1 shows a bar rack and a grit chamber as preliminary treatment techniques.

- Primary Treatment: focuses on the removal of settleable solids (typically organic), and substances such as fats, oils and grease that may float.

- Secondary Treatment: This is designed to remove residual organic matter and nutrients in wastewater using biological and chemical treatment processes. Aeration and clarification of mixed liquor are used to treat and reduce sludge.
• Tertiary Treatment: wastewater leaving secondary treatment can undergo treatment such as filtration and disinfection to achieve targeted removal of contaminants, such as pathogens and metals, alongside additional removal of suspended solids or nutrients.

For performance monitoring, the key areas of focus are the secondary and tertiary treatment stages. The objectives of secondary treatment on primary effluent wastewater are to reduce BOD and suspended solids of the wastewater effluent to acceptable levels and to remove nutrients [28]. Next, a tertiary treatment stage removes any residual suspended solids and is generally implemented by filtration through a granular medium or micro-screen. If not included at an earlier stage, disinfection and/or nutrient removal may form part of the tertiary treatment stage.

Further treatment is generally included when the effluent water is being considered for a particular reuse such as indirect potable reuse or when specific inorganic and organic constituents such as pharmaceuticals or heavy metals may be present. (Indirect Potable Reuse is the blending of advanced treated, recycled or reclaimed...
water into a natural water source that could be used for drinking (potable) water after further treatment).

### 1.2.1 Secondary Treatment – Activated Sludge Process

The activated sludge secondary treatment process, that treats wastewater through aerobic decomposition, is the most widely used suspended growth process, and broadly acknowledged as the most vital wastewater process [27]. Biological treatment occurs in the aeration tank where floc consisting of aerobic micro-organisms (bacteria, fungi, yeast, protozoa, and worms), particles, coagulants and impurities come together to form a biomass. This mass is generally irregular in shape and helps to collect pollutants (both organic and inorganic) in the wastewater by adsorption, absorption or entrapment. The aeration tank receives wastewater discharged from a primary settling tank or from a preliminary screening process and settled sludge is also returned to the aeration tank for reuse from the secondary settling tank. To maintain aerobic activity, air is pumped or blown into the aeration tank and the liquid, known as mixed liquor, is maintained in suspension by of mechanical mixing. The mixed liquor is transferred to the secondary settling tank (or clarifier) following a period of treatment in the aeration tank (generally 6-8 hours) [27].

The effectiveness of the operation in the aeration tank is considered the most critical aspect of the activated sludge treatment process [29] and its efficiency can be measured in a number of ways; most definitively by laboratory tests and often
implemented in practice by simpler, less accurate on-site tests.

In the secondary settling tank, the floc mass is separated from the liquid by sedimentation. The microscopic scale of single bacteria (0.5 – 1.0µm) does not allow for their removal by the physical process of sedimentation. However under conditions generated in the aeration tank, and to some extent the secondary settling tank, the bacteria form biomass floc that range in size from 10−70µm, [30,31]. As the biomass floc has a specific gravity slightly greater than water, gravity sedimentation can be used to separate the biomass from the treated liquid. Some of this biomass is returned from the secondary settling tank to the aeration tank for reuse (to maintain bacteria levels required for flocculation i.e. formation of particle clumps) and some of it is removed for sludge processing and disposal. Operators assessing the health of the activated sludge process will seek to determine whether oxygen transfer is occurring efficiently.

1.2.2 Tertiary Treatment

Wastewater treatment concludes with tertiary and/or advanced treatment. For example, treated wastewater leaving secondary treatment can undergo treatment such as filtration and disinfection to achieve targeted contaminant removal e.g. of pathogens and metals, alongside additional removal of suspended solids or nutrients.

Two common filtration methods are:

- Sand Filtration: Removal of suspended and colloidal solids from secondary
treated wastewater; typically used to achieve lower suspended solids and phosphorus concentrations in the discharged effluent.

- Membrane Filtration: Microfiltration or ultrafiltration processes combined with a suspended growth bioreactor.

### 1.3 Research Opportunities Targeted

#### Discharge Regulations

It is recognised that there is a need for innovative techniques that enable improved monitoring of WWTPs [32,33]. Furthermore such techniques must be simple and robust in order to lend themselves to real-time monitoring [34]. Examples of surrogate measurements include pH, oxidation-reduction potential (ORP), dissolved oxygen (DO), conductivity [35–37], modified SVI measurements [38] and Particle Size Distribution (PSD) determination. The key aim is to enable better WWTP performance and enable energy savings and efficiency in the WWTP sector [39]. There are many studies investigating the improvement of the biochemical oxygen demand (BOD), chemical oxygen demand (COD) and nutrient removal processes such as [40–42]. The Urban Wastewater Treatment Directive (UWWTD) defined the minimum standard required for wastewater effluent at 25:125:35 mg/l (BOD:COD:SS) [43]. However, this WWTP ”end of pipe” compliance monitoring is also regulated by the Water Framework Directive (WFD) ”ambient” monitoring programme for nat-
ural waterbodies. Therefore, individual WWTPs may have stricter discharge limits, based on the monitoring of the natural waterbodies in the area. Good communications between the compliance monitoring and enforcement authorities with the ambient monitoring programme is required. In reporting the results of monitoring there is a need for integration of programmes.[44]

The ability to correlate these metrics with the PSD could significantly improve operation of WWTPs, in terms of both efficiency and effluent quality. Measurement techniques for these standard metrics are time consuming - for example BOD is a 5-day test, while suspended solids (SS) and COD can both be measured using passive sensors; typically they are also measured using manual methods that can take several hours. Furthermore, in smaller plants such testing is often only carried out as prescribed by regulation; thus there is limited monitoring. PSD measurements can be instantaneous, can be used as accurate surrogates for BOD, SS and COD and thus could offer economical, accurate and robust methods to enable real-time or almost real-time monitoring and control. PSD based analyses have the potential to be very useful analogues for actual measurements [45].

1.4 Contributions

This thesis is concerned with the development of novel image processing-based measurements for performance monitoring of wastewater treatment plants. Improving the efficiency of operational monitoring at WWTPs requires the development
and implementation of appropriate performance metrics; particularly those that are quickly and easily measured, strongly correlate with WWTP performance, and can be easily automated, with a minimal amount of maintenance or intervention by human operators. These measurements will improve the current practices for WWTP monitoring by providing more regular measurements, and allowing for out-of-hours and remote monitoring. The technologies proposed may also have applications in other areas where similar metrics are used.

The key contributions of this thesis can be summarised as follows:

- A novel imaging technique to measure WWTP effluent turbidity.
- Measure of turbidity by manual observation of these images, through the use of a subjective test.
- An image processing system to objectively determine turbidity for these images.
- An automated system for measuring the stirred SSV of a mixed liquor sample wastewater sample as per Standard Methods 2710 C.

### 1.5 Thesis Structure

The remainder of this thesis is structured as follows: Chapter 2 reviews the current state-of-the-art in particulate measurements for mixed liquor and wastewater effluent. Existing methods are reviewed to identify gaps where automated measurement systems could facilitate more frequent performance monitoring. Chapter
3 details the system for determining the Settled Sludge Volume of mixed liquor samples. Both the image acquisition and subsequent image processing algorithm are described. Chapter 4 provides details for imaging of wastewater effluent for both subjective analysis of turbidity images and the dataset used for testing image processing-based turbidity measurement. In this chapter, the findings of the subjective rating of images of turbid effluent are also discussed. Chapter 5 outlines the objective measure of turbidity from these same images. Finally, Chapter 6 outlines conclusions, recommendations and proposed further research.

1.6 Publications to date

Four journal papers and one conference papers have been submitted for publication from this work:

Journal Papers


Chapter 1: Introduction


Conference Papers

Chapter 2

Literature Review of Sensor Technology in Wastewater Treatment

2.1 Introduction

Wastewater treatment is a significant contributor to global energy consumption (1 – 4%) [15, 46]. The most significant consumers of energy in a WWTP are plant control and operation functions i.e. pumping and aeration. There is also indirect energy usage associated with monitoring of the plant e.g. transportation of equipment and personnel. Control and monitoring of WWTPs is a key area of focus for plant managers as they try to meet discharge limits while minimising operational
costs. Wastewater treatment facilities are continually challenged to meet both environmental regulations and reduce running costs (particularly energy and staffing costs).

Regulatory bodies impose discharge limits on WWTPs [15, 47], and while these tests must be completed as per Standard Methods [48], the ability to monitor through surrogate measurements can enable the operator to maintain the WWTP within those discharge limits or take more prompt action when limits are breached between mandated regulatory reporting dates. In particular, for remotely located treatment plants, there can be significant inefficiencies associated with WWTP operation and maintenance. These inefficiencies are most often due to the fact that there is a lack of on-site staff to monitor performance, no provision for real time monitoring, with the result that performance is infrequently measured [49].

Korostynska et al. 2012 presented a review paper detailing the state of the art in water quality monitoring [50]. From this paper a number of key requirements for automated water monitoring can be identified:

- A suite of sensors for Sensor Fusion.
- Real-time monitoring.
- Innovative solutions to monitoring availability.

In activated sludge systems, the real-time or almost real-time measurement of activated sludge properties such as settling characteristics, particle shape and size
can be used to inform key aspects of plant control. BOD, SS, COD, nitrogen and phosphorous are the main performance metrics set out in binding EU legislation (Urban Wastewater Treatment Directive, 1991) [43].

The determination of particle size (PS), concentration and particle size distribution (PSD) (collectively known as the particle properties (PP)) in activated sludge and effluent wastewater samples has the potential to act as a surrogate measurement for multiple operational parameters at WWTPs. Much research has been conducted to establish the relationship between PS and PSD to other known performance metrics such as suspended solids concentrations and turbidity [45, 51]. PS and PSD can also impact on the operational efficiency of several treatment processes including: clarification, tertiary treatment, and disinfection. PS and PSD measurements, can be carried out rapidly, (unlike measurements such as chemical and biochemical oxygen demand) and thus have the potential to facilitate real-time monitoring of WWTPs, improve plant efficiency and thereby lead to cost effectiveness.

Several methodologies to determine the PP of wastewater samples are presented in literature. These vary from indirect methods such as laser diffraction to more direct laboratory analysis involving separation and filtration methods. However, there can be significant variation between these methods in terms of accuracy, measurement time and what parameters are being measured [52]. Such factors are important when considering a move towards real-time monitoring of WWTPs.

In this review, some of the metrics associated with sludge and effluent (secondary
and tertiary treatment) are outlined, along with the accepted standard measurement techniques. Also discussed is the practical considerations for on-site measurement methods for the same metrics. The importance of PP and PSD measurement in wastewater engineering, as well as surrogate measurements from the PSD are discussed in this thesis and examples of PSD measurement methodologies and their relative merits and demerits are examined. To date, reviews examining the methods and practices for the determination of the PSD with a view to characterising the performance of a wastewater treatment plant have been limited.

2.2 Metrics

There are two main areas of interest in a WWTP from a performance monitoring perspective: mixed liquor in the aeration tank and effluent from the plant.

As stated in Chapter 1, there are additional costs associated with the required monitoring of unmanned WWTPs, often in the form of indirect costs for transportation of equipment and personnel time. There has been ongoing research in wastewater assessment and monitoring over the last 20-30 years highlighting the need for low-cost, robust sensors capable of providing real-time feedback, enabling operators to make informed decisions [32, 43, 53, 54]. Furthermore, optimising the performance of WWTPs is a key area of focus for the sector in order to meet discharge limits set by EU legislation [7, 55, 56], as well as reducing operational costs [17, 57, 58]. The lack of full-time support staff often leads to infrequent measurement [6, 59] and a lack of
regular results for ongoing performance profiling.

2.2.1 Sludge Metrics - Secondary Treatment

The settling behaviour of activated sludge in WWTPs is a commonly used operational parameter used to assess performance and monitor changes in the wastewater treatment process. The settling characteristics of mixed liquor (sludge) have been shown to directly relate to the performance of the final settlement tank [60] and these settling characteristics are important for controlling the recycle rate for sludge to achieve the most efficient/effective treatment. Standard Methods for the Examination of Water and Wastewater Section 2710 [48] presents a series of tests designed for sludges or slurries. Results from these tests are used in the design of WWTPs for solids separation and concentration, and for assessing operational behaviour of the activated sludge process.

The measurement of plant performance metrics such as Settled Sludge Volume (SSV) and Mixed Liquor Suspended Solids (MLSS) (the two components of sludge volume index (SVI)), TSS, Nitrates and Phosphates are essential for efficient process control. The determination of SVI is key to describing the settling characteristics of sludge in the aeration process of wastewater treatment plants (WWTP). A major element of this calculation is determining the SSV, as well as the MLSS.

The SVI in particular has long been recognised as a useful index of plant operation with the parameter being frequently referenced in academic papers and technical
studies [38, 61, 62]. SSV is the volume of settled sludge in a 1L sample after 30 min. Slow settling is indicative of sludge bulking and poor compaction, while sludge that settles quickly leads to highly turbid supernatants in mixed liquor with a high suspended solids content [63]. A key aspect of any WWTP’s operational control is to ensure that the activated sludge settling characteristics are maintained within an acceptable range (often defined on a plant-by-plant basis[64]) to ensure discharge of a well clarified effluent.

MLSS is the mass of particulate contained in a mixed liquor sample used to measure SSV. MLSS is commonly used as a parameter for WWTP operation and control and at larger scale facilities, the application of PSD has been used to investigate settling problems, characterise sludge and inform managers on the overall health of their WWTP. The gravimetric determination of MLSS, which is used as the standard to calibrate and check other direct suspended solids concentration measurement devices e.g. MLSS probes, has potential to also provide PSD information through the application of serial filtration.

**Standard Methods for determining SVI**

The ratio of SSV to MLSS is the SVI [26], as shown in Equation 2.1

\[
SVI, mL/g = ((SSV, mL/L) \times (10^3, mg/g))/(MLSS, mg/L) \tag{2.1}
\]

Current practices for calculating SSV are described in Standard Methods 2710 C
Figure 2.1: Mixed Liquor sample settling in a clear vessel. A floating scum layer can be seen at the top of the vessel and a non-uniform supernatant-sludge interface forms as the mixed liquor sample settles.
The SSV is measured as follows: A mixed-liquor sample is placed in a 1 L cylindrical vessel. The sample is allowed to settle for 30 minutes and the volume of the settled sludge i.e. the SSV, is measured; this value is commonly referred to as SSV30. MLSS measurements are found by serial filtration of a mixed liquor sample and subsequently obtaining the dry mass of particulate matter present.

**Modified SVI measurements**

There are variations on the SVI, also outlined by Standard Methods. Depending on the sludge concentration, it is possible to have a SSV value close or equal to 1000 mL/L, in this case the sample would not have settled after 30 minutes. In such a case, it is recommended [65] that a diluted version of the test is used whereby the mixed-liquor is repeatedly diluted until the SSV result is below 250 mL/L [48]. Where a settlement test is being carried out on a diluted sample, the required dilution is achieved by the addition of water to a known volume of the mixed liquor sample to achieve a 1:1, 1:2, 1:4 or 1:8 dilution followed by brief manual stirring to ensure homogeneity. SSV is then scaled as shown in Equation 2.2, where \( n \) = number of two-fold dilutions.

\[
DiluteSSV, mL/L = ((SSV, mL/L) \times 2^n)
\]  

(2.2)

The SVI is then computed as per 2.1. Additionally, a stirring device can be used to aid settling by minimising wall effects on the settling solids; this is known as the stirred SVI [26, 48]. Stirred SVI is a key area of research for this thesis. Stirred SVI
measurement replicates settling in the secondary clarifier and therefore indicates the likely sedimentation of biomass [48].

**On-site Methods**

While the internationally recognised Standard Methods [48] outline the procedure for measuring SSV (a component of the sludge volume index (SVI)), in practice this is not always adhered to for day-to-day monitoring. There are large variations in the SSV measurement technique used by different plant operators and caretakers, especially in small-scale remotely located WWTPs [38]. A variation of the SSV measurement, known as the cone-test, is widely used as an accepted proxy measurement of plant performance [16,38,66].

While the measurement procedure for SSV is clearly defined, in practice, a wide variety of vessel sizes and shapes are used by plant operators to monitor performance, with operational procedures often defined on a plant-by-plant basis. This leads to a large SSV variance and non-comparable performance metrics between WWTPs [67]. In an Irish context (and likely repeated elsewhere) it was found that a variation of the volumetric settleable solids test is the only sludge settlement test regularly performed at most WWTPs with test frequency varying from daily to weekly [38]. Most commonly, the test comprised a 30-min period of quiescent settling in an Imhoff cone. The volume of sludge settled was then recorded in place of an SSV measurement [64]. However, this cone test result is not comparable to the internationally accepted
Chapter 2: Literature Review of Sensor Technology in Wastewater Treatment

Standard Methods stirred SSV test [48] that is used along with MLSS to derive the SVI parameter. Furthermore, the settleable solids test, on which the cone test is based, is recommended for dilute sludges, which are frequently not representative of activated sludge processes and thus the cone test, despite its widespread use, has been shown to be of limited value [38].

Furthermore, the study by Hannon et al (2014) indicated that settled volume measured from an unstirred settlement test is dependent on the vessel type, i.e. shape, in which the settling is taking place and that the settled volumes derived from unstirred settling in a 1 L graduated cylinder and 1 L Imhoff cone were consistently higher and showed less variation than those derived from settleometer vessels. This is an important finding as the results presented also found that the Imhoff cone is the most commonly used vessel in operational testing.

Inconsistencies in SVI measurement techniques developed from guidelines in The Waste Water Treatment Manuals themselves [68]. These manuals set out the general principles and practices that should be followed for treatment of municipal wastewater. They provide practical guidance to those involved in plant operation, management, maintenance and supervision, and recommend that a management system be developed to ensure that the objectives of the plants in terms of treatment are achieved. The manual correctly defines SVI as the metric to assess the settling qualities of sludges. However, the method for measuring SVI is not in full accordance with Standard Methods [48]. Instead, the manuals describe SSV measurement as ‘filling
an Imhoff cone (or 1 L graduated cylinder) with mixed liquor from the aeration tank and allowing the mixture to settle for 30 minutes.’ The volume of settled sludge (in 1 L) is read after this period and the SVI is computed. The manual defines that a sludge which settles well has an SVI value of 80 mL/g or less and values as low as 50 mL/g indicate a very good settling sludge whereas values greater than 120 mL/g indicate poor settling. This range is roughly in-line with internationally accepted results [26], that indicate an SVI of 100 mL/g or below is considered indicative of good settling sludge and SVI values above 150 mL/g are generally associated with filamentous growth of sludge. Poor settling leads to inefficient treatment, and increased energy consumption.

The EPA manual concludes that for routine monitoring of sludge settleability, the SVI test is adequate; however it notes the following:

- The described test is quiescent and as such, the dynamic hydraulic conditions in the settlement tank are not reflected.

- At MLSS concentrations in excess of 2500 mg/L, wall effects (fluid dynamics with the vessel structure) may occur in the measuring cylinder, distorting the rate of settlement.

To overcome these problems, the manual recommends using the stirred sludge volume index (SSVI). This test is equivalent to Standard Methods test 2710 D.
Summary of sludge metrics

Treatment metrics for the thick sludge-like composition of mixed-liquor include SVI, MLSS and SSV, each requiring time-consuming measurements that are performed both on and off site at a significant cost. Although, the importance of activated sludge settlement and its measurement as an indicator of the performance of the activated sludge process is well recognised, the variations in test methodologies, referenced both in literature and standards, shows little consistency of approach to determining settlement characteristics. The study by Hannon et al [38] indicates that settled volume found from an unstirred settlement test is dependent on the shape of the vessel in which the settling is taking place, and that the settled volumes derived from unstirred settling in a 1 L graduated cylinder and 1 L Imhoff cone were consistently higher and showed less variation than those derived from settleometer vessels. This variation in SSV between vessel types is an important finding as the results presented also showed that the Imhoff cone is the most commonly used vessel in operational testing in Ireland, even though the Imhoff cone test is not representative of settling in the clarifier.

It is clear that, in this area there is scope for the development of automated systems that conform to the guidelines set in Standard Methods, both to improve measurement practices and allow for more frequent measurement. In future, these automated systems could then be integrated with a centralised data management system to perform autonomously, reporting and recording performance parameters
for both operational and compliance monitoring. Measuring the SSV can also give an early indication of effluent quality.

It is evident that a device/system capable of calculating SSV as per the Standard Methods would help improve both operational monitoring as well as compliance reporting. The work presented in the following chapters identifies the commonly used settlement assessment techniques carried out at activated sludge WWTPs and establishes relationships between results derived from these methodologies and proposed new image-based metrics.

2.2.2 Effluent Quality Metrics - Tertiary Treatment

Effluent should have a much lower biomass content than mixed liquor. The visual performance metrics associated with effluent include TSS and turbidity. There are also other, non-visual metrics associated with dissolved organic matter, such as pH, BOD, COD, nitrate and phosphate levels. The EPA and EU enforce strict guidelines as to the levels of nitrogen, phosphorus, BOD and COD as well as particulate matter that is acceptable to be released from a WWTP. There are correlations between these physical metrics, TSS and turbidity, and the chemical metrics such as COD.

Multiple studies have examined the chemical composition of wastewater effluent samples [69–71]. The chemical composition of the particulate matter is important for assessing effluent quality. Azema et al. 2002 presents a graph of the chemical composition of particle size ranges in a number of different wastewater types. The
main constituents of the 4 particle groups typically found in effluent are shown in Table 2.1 [72]. An effluent sample was allowed settle in an Imhoff cone with a capacity of one litre. The resulting settled sample was then divided into 4 parts volumetrically using gravimetric settling, and subsequent particle size measurement was conducted using laser diffraction.

Table 2.1: Major components of particle groups. Adapted from (Azema et al. 2002).

<table>
<thead>
<tr>
<th>Particle Group</th>
<th>Chemical Composition</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dissolved</td>
<td>Nutrients, Fulvic &amp; Humic Acids, Carbohydrates, Amino-acids, Vitamins, Fatty Acids.</td>
</tr>
<tr>
<td>Colloidal</td>
<td>Cell Fragments, Virus, Cellulose, Pectin, Proteins, Exocellular Enzymes, Bacteria.</td>
</tr>
<tr>
<td>Supracolloidal</td>
<td>Algae, protozoa, Bacteria, Bacterial Flocs, Organic debris</td>
</tr>
<tr>
<td>Settleable</td>
<td>Bacterial Flocs, Organic debris</td>
</tr>
</tbody>
</table>

The results showed that the first three volumetric fractions of the effluent were mainly composed of supracolloidal particles in a single mode distribution centred on 30µm, while the final fraction (the lowest point of the inverted cone) contained a multimodal distribution of particles greater than 100µm in size. Contrastingly, another study focussed on the concentrations and compositions of proteins, carbohydrates and lipids in wastewater samples at various stages of the treatment processes for municipal, industrial and agricultural treatment plants [73]. Both studies used
particle measurement techniques to characterise the structure of wastewater particles i.e. mass, volume, density. The use of ultra-violet (UV) spectrophotometry can also allow for further highlighting of the contributors to COD and SS [72]. These studies establish the use of PSD as a surrogate measurement for other wastewater parameters. Further studies have investigated other surrogate measurements, highlighting the importance of the PSD in the trends of DO, ammonium and nitrate concentrations [74]. There is a need for improvements in systems and techniques for determining PSD in the wastewater treatment field to help establish the use of PSD as a surrogate measurement for other metrics. This would create an avenue for implementing a real time remote control and monitoring solution, as such developments will be required in the future to effectively manage the many remotely located small-to-medium scale plants [75–79].

**Turbidity**

Turbidity is one of the simplest measurements associated with suspended particles in a fluid; it can be easily equated with the cloudiness of the fluid which is governed by the number of small particles present. Turbidity is widely used as a general indication of performance in water treatment systems as a measure of effluent clarity, and studies have found it can also be used as a surrogate measurement for Total Suspended Solids and Chemical Oxygen Demand [80–82]. Turbidity is primarily affected by the smaller particles that generally dominate frequency distributions [83]. Tests have shown that
turbidity increases during the anaerobic phase and decreases again during the aerobic phase of wastewater treatment[61].

While turbidity measurements are not included in the compliance monitoring guidelines imposed by the European Council [43], turbidity can be a useful surrogate for other performance parameters. Cloudy effluent samples often indicate poor process control or a problem within the treatment system; e.g. poor settling in the clarifier could be indicated by a higher turbidity measurement in the effluent. When determining SSV, the turbidity of the supernatant left after the sludge has settled can also be indicative of the effluent turbidity [64].

Total suspended solids (TSS) of wastewater effluent, a measure of the mass of particulate matter in a fluid, has also been found to correlate with turbidity[12,25,84–87]. TSS is a significant wastewater effluent quality metric and a good indicator of other pollutants, particularly nutrients and metals that are carried on the surface of sediment in suspension [84]. An increase in effluent turbidity generally indicates an increase in TSS. Using linear regression and site-specific calibration, an estimate of TSS (normally a labour-intensive measurement) can be made from a relatively quick and easy turbidity measurement [82]. This relationship between turbidity and other wastewater effluent properties such as particle size distribution (PSD) can provide useful information about WWTPs for performance monitoring. PSD has been found to have a significant influence on turbidity as well as TSS and COD [83,88,89]. The latter study by Wu et al highlights the inverse relationship between particle size and
turbidity. On the other hand, TSS was also found to be highly correlated with the volume of particles, with the larger particles being the biggest contributor to volume. Furthermore, multiple studies have also established the relationship between infrared spectroscopy and COD, BOD and TSS [22–24]. Building upon these relationships a turbidity monitoring system could provide useful information on WWTP performance.

New sensor technologies are constantly being developed for surrogate monitoring of performance metrics in the wastewater treatment process [90–92]. These sensors measure standard performance parameters such as COD, BOD, DO, phosphates and ammonia. Online sensors have also become prevalent in other aspects of the treatment process, such as optimising external carbon dosage and measuring nitrous oxide emissions [93,94]. The primary advantage of in-situ sensors is the rapidity of the measurement in comparison with laborious standard practices, often involving laboratory equipment, multiple stages of sample process and analysis, expensive and sensitive equipment [95]. However, existing sensor systems are generally only reliable under specific conditions such as; a stable organic load and chemical parameters, making them unsuitable for long term monitoring of diversified samples. Bourgeois et. al presents a comprehensive review of the limitations of these types of sensor [21]. Fouling and location dependency of submerged sensors have been highlighted as the most significant issues [19,20]. There is a need to look at new ways of proxy monitoring through the development of more sensors. Specifically, previous studies
have found that optical effluent monitoring systems can be used to predict suspended solids measurements in biologically treated wastewater [96,97].

**Other applications of turbidity outside wastewater treatment**

In drinking water treatment systems, the presence of dissolved organic matter (which are light-absorbing substances) such as tannins can cause discoloration and cloudiness of the water supply which can indicate poor water quality. These dissolved substances may be too small to be counted in a suspended solids measurement, but they do contribute to turbidity as they affect water clarity. From a drinking water quality perspective, turbidity is not an accurate quality metric; however, it is an important metric for treatment plant efficiency [98]. While dissolved matter is not included in TSS measurements, it can cause artificially low turbidity readings as it absorbs light instead of scattering it [53,99]. Turbidity has also been found to affect the efficiency of chlorination in drinking water disinfection. Particulates in drinking water supplies can act as carriers for bacteria, heavy metals, and pesticides. While turbidity does not provide a direct measure of the levels of these other potentially harmful impurities, its presence highlights the fact that a viable mechanism exists for carrying them into the drinking water supply [100–102]. Turbidity monitoring requirements, as specified in the drinking water treatment directive, state that a minimum error of 12.5% is acceptable for the measurement of turbidity [103,104].

Turbidity is also an important factor in sectors other than water treatment. Some
marine applications of turbidity include: using turbidity to examine the environmental effects of dredging [105,106] and the effects of turbidity on fish and other aquatic life [107–109]. Additionally, turbidity is also utilised in the pharmaceutical industry to monitor the rate of emulsification of drugs [110].

**Turbidimeters**

In-line devices suffer from accuracy issues over time, due to the challenging environmental conditions in WWTPs i.e. nutrient rich, moist growing medium. However, with regular maintenance, these devices have been shown to perform well [79].

A number of systems are currently available for estimating fluid turbidity, ranging from fully automated systems that measure the attenuation of light in a fluid such as spectrophotometers or turbidimeters, to more subjective measurements of the depth of fluid through which a target can no longer be seen, such as the Secchi Disc method [111]. Laboratory based turbidity measurements are generally performed using a spectrophotometer or dedicated turbidimeters and require manual calibration. Turbidity is calculated by comparing the light attenuation of the sample to that of deionised water [112].

Turbidimeters are generally deployed on-site at treatment plants, but prolonged submersion in the turbid fluids can lead to build up of organic material on the sensors, reducing their accuracy. Current automated solutions require regular cleaning and calibration as the sensor fouls due to the build-up of particulate matter; this is
particularly an issue in the wastewater sector [113]. Regular maintenance of this type of sensor is required to ensure the readings remain reliable [79]. The majority of these systems are based on spectroscopy with a light source at a fixed wavelength (predominantly 860nm) and a paired detector that measures the intensity of light scattered towards it [89,114]. Recently, a number of lab-based systems have been developed based on lasers and opto-electronics [115]. The new opto-electronic based systems are smaller, easier to use and more convenient devices that can be used to measure turbidity, particularly in developing countries, where fresh water is often a scarce resource. A pilot-scale automated system that monitors treated water by measuring the refractive index of the effluent stream has been tested in small-scale applications and demonstrated the ability to detect water quality issues such as turbidity in a water distribution network [75,79].

While it may be feasible to deploy existing automated systems in larger plants with technicians onsite regularly, this is not always possible in small to medium-scale treatment plants where monitoring is less frequent. Therefore, due to a lack of calibration, results from submerged turbidity sensors are rarely deemed trustworthy after more than a few days. However, in ideal conditions, where the sensors are regularly cleaned and calibrated, they can yield a reliable and accurate estimate of turbidity.

Regulatory reporting only being required monthly or a few times per annum, often leads to minimal maintenance of monitoring systems in WWTPs. There are many
issues with data management and calibration of monitoring and analysis systems in WWTPs. It is a costly, manual task in terms of operator-hours and audits have found frequent instances of non-compliance with regulations [16]. Unfortunately, lack of calibration can quickly lead to unreliable results. Currently, subjective visual analysis methods for WWTP effluent on-site are somewhat informal and irregular. There are financial penalties associated with non-compliance with discharge limits. The introduction of automated – remotely operated performance sensors could significantly reduce the costs associated with plant monitoring by reducing the requirement for on-site monitoring, particularly in cases where WWTPs are operating effectively and complying with their respective discharge limits.

2.3 Particle Size Distribution

2.3.1 Settleability and Particle Size

Metrics associated with particles including PS, PSD and concentration are important parameters that can provide valuable information on wastewater composition and WWTP performance. It is widely accepted that knowledge of the particle size distribution of activated sludge, can lead to a more complete understanding of the activated sludge process and its control [116]. In particular, the size of the sludge flocs is an important indicator of the settling properties of activated sludge [117]. Multiple studies have investigated the particle properties (PP) of wastewater in-
fluents and effluents \[40, 88, 118-121\] as well as in bioreactors \[122, 123\], and the findings of these studies highlight the relationship between PP and several key parameters of treatment processes, each of which are discussed throughout this review. To understand the significance of PSD in wastewater treatment applications, it is necessary to characterise the size of particulate matter present in wastewater. Four particle size ranges are most commonly identified in the literature, as shown in Table 2.2,\[70,72,124\]. Particles generally range from \(< 0.001\mu m\) to over \(100\mu m\) in diameter in settled wastewater \[72,73\].

<table>
<thead>
<tr>
<th>Particle Group</th>
<th>Size Range</th>
<th>Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td>Soluble</td>
<td>(&lt; 0.001\mu m)</td>
<td>Largest contributor to COD &amp; BOD.</td>
</tr>
<tr>
<td>Colloidal</td>
<td>(0.001 - 1\mu m)</td>
<td>80% of all effluent particles are in this group, turbidity is largely affected by number of particles</td>
</tr>
<tr>
<td>Supracolloidal</td>
<td>(1 - 100\mu m)</td>
<td>Mixed liquor PSD usually bimodal, centre in this region, particle shape can affect settling (SVI).</td>
</tr>
<tr>
<td>Settleable</td>
<td>(&gt; 100\mu m)</td>
<td>Large mass/volume contribution to SS.</td>
</tr>
</tbody>
</table>
2.3.2 Relation of PSD to wastewater characteristics

The main goal of determining the PSD of wastewater is to use this relatively simple metric as a surrogate for other, more complex characteristics such as suspended solids and COD. The PSD of an effluent sample is an extremely important characteristic in the evaluation of tertiary treatment methods; these tertiary methods include the removal of fine particles from the effluent stream [80,125–129]. Table 2.3 outlines the relationship between various wastewater and sludge characteristics and particle size.

While components from all four groups are generally present to some degree in all wastewater samples, the relative importance of each group varies when attempting to use PSD as a surrogate measurement. For example; the turbidity of the supernatant is affected more by smaller colloidal particles; however, colloidal particles are minor contributors to overall SS mass compared to supracolloidal and settleable particles [83]. Therefore, colloidal particles could be disregarded in a MLSS estimation. However, if the constituent organic matter is being examined, studies have found that a considerable percentage is contributed by both the soluble and colloidal groups [70,130].

While 80% of all particles (as measured by particle numbers) are in the $2\mu m – 6.8\mu m$ range [121], many PP measurement techniques are more suited to specific size ranges. Given the wide range of particle sizes generally present in wastewater, multiple measurement methods are often required to accurately determine the full
Both particle size and shape greatly affect the settled sludge volume (SSV), and the sludge volume index (SVI) of mixed liquor in WWTP clarifiers. Due to the infrequency with which these tests are conducted, PSD could provide a useful surrogate measurement for these parameters when compared with other commercially available solutions for SVI, as it can detect sludge bulking events that can lead to an increased SVI [132]. While earlier studies argued that the main contributors to BOD and SS are particles less than 50µm [73], recent studies indicate that particles significantly larger than 50µm are the main contributors to SS [61, 83, 133]. The majority of reviewed literature focuses on the supracolloidal and settleable SS group, 80-95% of the total particle volume is occupied by particles greater than 50µm [83]; these particles have the biggest impact on mixed liquor settability and MLSS, while particles smaller than 50µm are the largest contributors to effluent turbidity.

The impacts of dissolved oxygen (DO) on PSD of activated sludge WWTPs and wastewater effluent has been examined in detail [42, 61, 134]. In these studies, both the settling properties of activated sludge and the turbidity were compared for samples exposed to various aerobic and anaerobic conditions (thus varying DO). Due to the large size range over which particles needed to be measured, three different measurement techniques were employed: a manual microscopic analysis and two laser diffraction systems. It was found that no consistent mathematical correlation could be found between DO concentrations and particle size. However, it was observed
Table 2.3: Relation of wastewater performance metrics to particle properties

<table>
<thead>
<tr>
<th>Metric</th>
<th>Relationship with particle properties</th>
</tr>
</thead>
<tbody>
<tr>
<td>SVI</td>
<td>Filamentous particles do not compact and therefore settle poorly.</td>
</tr>
<tr>
<td>SS</td>
<td>Particles &gt;50µm are the largest contributors to SS.</td>
</tr>
<tr>
<td>COD/BOD</td>
<td>Mostly contained in the soluble fraction &lt; 0.001µm.</td>
</tr>
<tr>
<td>DO</td>
<td>No direct correlation, but larger denser flocs at higher DO concentration.</td>
</tr>
<tr>
<td>High Turbidity</td>
<td>Indication of higher number of colloidal particles.</td>
</tr>
<tr>
<td>Bulking Sludge</td>
<td>Compacts poorly, RAS increase, Filamentous Particles.</td>
</tr>
</tbody>
</table>

that there was a tendency towards the development of larger flocs at higher DO concentrations; these large flocs were also more compacted than those at lower DO concentrations [61]. Alternating anaerobic/aerobic phases had little effect on the settling properties; however, a change in turbidity was observed. Subsequent analysis showed a decrease in the number of small particles during the anaerobic phase.

2.3.3 Particle Size Distributions

A variety of PP results are presented in the literature, with the format of the results depending on the method of measurement used. Results are most commonly presented as either a frequency of occurrence distribution or a mass/volume distribution. Frequency of occurrence distributions for a wastewater sample are generally centred around 1 µm range as these are the most common particle sizes present in a
mixed liquor sample. A size distribution however, generally are centred well above 50 µm as these are the largest contributors to the size (mass/volume/surface area).

Particle characteristics and their relationship to measurement techniques will now be discussed, highlighting the components of the distribution that contribute to the wastewater characteristics (BOD, SS, SVI) and how treatment processes can affect the distribution.

Weighted Mean

The method by which the particles are measured can often influence how results are presented. For example, analysing a particle under a microscope yields a two-dimensional image. Meanwhile analysing particles using laser diffraction gives a range of different equivalent diameters, the most common being the volume moment mean, denoted D[4,3], which is a type of weighted mean calculated based on the quotient of multiple powers of the measurement, shown below in Equation 2.4.

The measured diameter of each particle can be averaged to provide a single equivalent diameter; one such example is the use of weighted means. The $n^{th}$ order of this mean varies, a first order average (length based mean) is the standard mean diameter, where the sum of the diameters is divided by $n$, the number of particles. This model applies to higher order means also (area, volume), shown in Equation 2.3:

$$D[a,0] = (\Sigma d^n)/n$$  \hspace{1cm} (2.3)
Chapter 2: Literature Review of Sensor Technology in Wastewater Treatment

where $D$ is the mean particle size, $n$ is the number of particles, $a$ denotes the order of the mean i.e. 1 is length mean, 2 is area mean, 3 is volume mean. This weighted mean is a useful metric, as it provides a conversion from a frequency distribution to a mass/volume based average; it is a common metric for comparison with direct methods such as serial filtration.

The volume moment mean $D[4,3]$, shown in (2.4) is reported by numerous sources with the results usually generated by laser diffraction systems such as the Malvern Mastersizer [83, 135]. The volume moment mean is calculated as the sum of 4th powers of the particle diameters divided by the sum of the cubes of the diameters. One advantage of this volume moment mean is that the number of particles is not required, as $n$ cancels above and below the line, as seen in (2.4,2.5).

$$D[4,3] = \frac{\sum d^4}{\sum d^3} \quad (2.4)$$

$$D[3,2] = \frac{\sum d^3}{\sum d^2} \quad (2.5)$$

The surface area moment mean (denoted $D[3,2]$ shown in (2.5) also known as the Sauter diameter) is defined as the diameter of a sphere with the same volume/surface area ratio as the particle of interest (Rodrigues and Rubio, 2003). This mean is an interesting metric as it provides a representation of the ideal system where all particles are uniform spheres. Comparing this measurement with the elliptical/filamentous nature of the particles can yield results relating to the shape of
the particles. Particle shape is an important factor in mixed liquor settling.

**Particle Size and Frequency Distributions**

Early work on assessing the PSD of activated sludge found it to be a bimodal distribution with one group of particles in the range of 0.5–5 μm and another group in 25–2,000 μm range [136]. Due to the high settling velocity of larger particles, a very high removal rate (up to 100%) of large particles was achieved, while the percentage of mass removal for the small size group was lower due to their lower settling velocity. The work of Dahong & Ganczarczyk (1991) on floc sizes similarly concluded that for the samples analysed, while particles less than 5 μm were dominant in number, those larger than 50 μm contributed most surface area, volume and mass. This size analysis was based on activated sludge samples from five conventional WWTPs where a Coulter counter was used for particles less than 10 μm and individual image analysis for particles of larger sizes.

Due to the range of particle sizes present, log-normal scales are utilised to represent the distribution graphically [116]. There are differing results on the spread of bimodal distribution analyses. Multiple sources found few flocs in activated sludge from WWTPs between 5 and 25 μm [116,137], while more recent studies have found that samples are dominated in number (80%) by particles between 1–10 μm; in terms of volume however the majority contribution comes from the 80-100 μm range [83]. This contrasts with previous studies that claim that the material which comprises
the SS of settled municipal wastewater is usually smaller than 50µm. [73]. However, these studies used different measurement techniques and potentially made different assumptions about particle shape and density; therefore, they are not directly comparable. This disagreement in the results highlights a need for a standardisation of particle measurement techniques and the need for a review of the impact of measurement method on results in wastewater assessment.

For MLSS analysis, it has been argued that any particles smaller than 25µm can be ignored for calculation purposes due to the lack of particles in the 5 to 25 µm region [116] and the fact that the majority contribution to volume comes from larger particles [83]. However, previous studies have found that particles in this range (5 to 25 µm) have a significant impact on the chemical and biochemical characteristics of wastewater. For example, one study of the chemical composition of primary effluent samples from municipal WWTPs examined the distribution of particles in this range and observed that only 45% of the COD present in the sample was represented by particles greater than 1.2 µm [70].

2.4 Common methods for PSD determination

Table 2.4 summarises the methods predominantly used to determine particle size distribution for both mixed liquor and effluent samples in wastewater treatment. A number of studies have examined the structure and physical characteristics of sludge flocs and concluded that laser diffraction is the preferred method of determining the
particle size distribution (PSD) of activated sludges [138, 139].

<table>
<thead>
<tr>
<th>Methods</th>
<th>Range</th>
<th>Notes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Laser Diffraction</td>
<td>0.005µm – 100µm+</td>
<td>Multiple variables, no standard for comparison between studies</td>
</tr>
<tr>
<td>Filtration</td>
<td>0.05nm – 200µm+</td>
<td>Massive range, labour intensive, long wait times</td>
</tr>
<tr>
<td>Microscopy</td>
<td>0.1µm – 200µm+</td>
<td>2D analysis of 3D objects, shape and density must be assumed</td>
</tr>
<tr>
<td>Sedimentation</td>
<td>0.5µm – 150µm+</td>
<td>Often used for first pass separation before subsequent analysis</td>
</tr>
</tbody>
</table>

2.4.1 Laser Diffraction

Description

The Malvern Mastersizer [140] is the device predominantly used in the literature for laser diffraction, although there are alternative solutions from other manufacturers: e.g. Coulter, Retsch and Cilas. The PSD estimated using this method is volume weighted rather than by frequency of occurrence. While the use of laser diffraction is frequently observed in studies discussing PSD, [69, 141–145], multiple sources argue that this method does not yield an accurate representation of the actual PSD in the
sample [133,146], but rather can be used as a relative metric after the standardisation of measurement techniques.

**Background**

There are some points worth highlighting with laser diffraction analysis:

A study examined the stability of the medians and deciles of the PSD floc after repetitive measurements (i.e. the same sample passing through the laser diffraction system twenty times) [133]. Examination of the mean was not conducted because of the expectation that the intense stirring would disintegrate particles. The results found that floc size was affected by pump and stirrer speed in the unit. Larger flocs were fragmented by the pump and stirrer more than smaller flocs. The median particle size decreased with increased rotor speed. Finally it was found that after the repeated measurement (20 times) the results had stabilised. This study raises questions about the reliability of laser diffraction analysis as a direct measurement of particle size distribution. While on one side it ensures test repeatability, it is not an accurate representation of the sample when it was originally taken from the plant. The prolonged stirring caused some break down of particles/flocs. Obviously, no mass has been lost from the system, so it would yield the same suspended solids measurements, however the PSD had a lower number of larger particles. Rather than using laser diffraction for a direct characterisation of the PSD of the sludge, the results could be used for comparative studies. One major point to note from
these comparisons is that in the use of laser diffraction, the results do not provide a real measurement, merely a relative metric. Use of this metric is valid provided there is some standardisation of measurement techniques and a clear indication of assumptions made for the relevant particle theory (Fraunhofer or Mie) [147]. The problems with these theories for wastewater applications, primarily Mie, is the assumption that all particles are spherical; however, microscopic studies have shown that this is not the case for wastewater [148,149].

**Observations**

Laser diffraction is one of the most widely used methods for determining PSD, as shown in recent literature. Such systems are readily available as commercial solutions. Under standardised conditions, laser diffraction can provide a relative metric of PSD. It should be noted however that variations exist in the measurement procedures and equipment used. Studies have found a lack of information regarding the assumptions made and the measurement procedures used in laser diffraction analysis, questioning the validity of the comparison of these results [133]. For example, there are two common algorithms that convert light intensity to particle size: Mie and its subset Fraunhofer [147]. Knowledge of the assumptions (refractive index and absorption coefficient) made in these algorithms is required for comparing the results. If these assumptions are not clearly stated, no definitive comparison can be made [145]. In some cases, large discrepancies in the results presented from different
laser diffraction devices have been found [146].

2.4.2 Filtration

There are a limited number of studies in the literature that discuss direct PSD measurement methods, the majority of which involves mechanical separation of particles through filtration [69, 150, 151]. Serial Filtration is the most common technique employed in the literature reviewed.

Description

Serial filtration involves the use of progressively finer filters or mesh sizes to characterise the mass distribution of filterable solids [152]. Common sized mesh sieves (63µm, 38µm) are used for a first pass separation, subsequently membrane filters are used for microfiltration (10µm, 1.2µm & 0.2µm) of particles in the colloidal to supracolloidal range. The filters are prepared as outlined in the Standard Methods [48]. Further filtration can be conducted on particles smaller than 0.2µm using ultrafiltration membranes (1000amu, 100000amu). Sophonsiri et al. 2004 used filtration to investigate the chemical composition of the four particle size ranges of activated sludge [70]. While the paper did not define a PSD directly, it is a useful insight into the filtration methods available. Both parallel and serial filtration methods are employed where necessary.
Observations

Serial Filtration is one of the most basic methods for determining PSD. The process is easily repeatable and not laborious. Drying time for the filter papers is usually in the order of hours in a low temperature oven. Thus, the results are not instant and hence this method is not ideal for pseudo-real-time performance monitoring.

Serial filtration utilises equipment commonly available at many WWTPs, making determination of PSD relatively achievable for plants without ready access to laser diffraction equipment. The effect of laser diffraction tests on particle size in activated sludge samples is significant [133], therefore serial filtration may offer significant benefits over laser diffraction in terms of providing PSD analysis with easily interpretable results, without the necessity for sophisticated laboratory apparatus.

2.4.3 Microscopy

Description

Microscopy-based PSD analysis involves placing a sample under a microscope for analysis. The analysis methods vary from the simple user drawn particle size (i.e. the operator highlights the particles using a light pen) [153] to more complex automated image processing systems, using software for analysis of microscopic images to determine particle sizes [148]. Multiple studies investigate the morphology of activated sludge particles [148, 154, 155].
Numerous literary sources utilise microscopic imaging systems for wastewater analysis, [61,116,144,151] though not always to determine the PSD [148]. While the latter study does not present the results in the form of a PSD, it employs the same techniques normally used to determine PSD with microscopy. The aim was not to provide a PSD; instead to estimate the “settlesability” and concentration of activated sludge. The study uses form factor (comparison of the roughness of the particle compared to a circle) and aspect ratio (related to the elongation of the particle). As they were only concerned with flocs larger than 10µm, there is a practical limit to the size of particles for which this methodology is suitable. The contribution of particles smaller than 10µm were also deemed inconsequential to sludge settleability. This study of sludge morphology highlights the possibility of estimating the concentration of activated sludge in the system; another potential surrogate measurement.

Observations

The availability of microscopes in the majority of laboratories allows for a simple analysis to be performed promptly. The porous nature of activated sludge flocs makes them difficult to accurately characterise by 2D analysis; therefore, some assumptions must be made about particle shape (spherical, ellipsoidal, disc) and their density to define a particle mass.

In an ideal situation, all particles would be spherical in nature and therefore (assuming a density) could be described by a single number: the diameter. However, in
practice, it has been found that particles come in a variety of shapes: ellipsoidal, discs and filamentous. To address the lack of a 3-dimensional diameter value, for practical reasons an approximation is necessary, and there are several potential equivalent diameters that could be implemented. Examples of alternative diameter definitions include: maximum, minimum and Feret (diameter between two parallel planes, often called calliper diameter), as well as surface areas that can be converted to an equivalent sphere [155] for comparing particle distributions in samples. However, it is important that the equivalent diameter dimension adopted is applied consistently between the results being compared[156].

2.4.4 Sedimentation

The simplest method for separating particles in a water solution is to allow them to settle; larger particles generally settle to the bottom [118,157]. The separation is based on Stokes Law and is dependent on the diameter and density of the particles. Sedimentation does not generally provide in-depth results, but can be useful for a first pass separation to facilitate subsequent analysis, such as examining the primary particle size contribution to TSS in wastewater [72]. This method is crude and simple and is often followed by subsequent analysis using one of the other methodologies previously detailed in this section.
Chapter 2: Literature Review of Sensor Technology in Wastewater Treatment

2.5 Summary and Conclusions

This chapter discusses the suitability and reliability of a variety of techniques to determine the PSD of wastewater samples. From the limited number of studies in the literature that have examined the impact of PSD on key performance metrics, it was found that the PSD had significant influence on these metrics. In relation to these PSD measurement methods, a number of questions must be addressed:

- Are they suitable for on site deployment or in a laboratory?
- Are they accurate and repeatable?
- What is their ease of use in practice?
- Can they be used as surrogate measurements?

Commercially available laser diffraction systems can be deployed on site, and while these systems are a viable option for larger urban treatment plants, for small plants they may not be financially appropriate. Lab based methods require a variety of specialised equipment and qualified technicians; hence these solutions do not always lend themselves to on-site use due to the capital cost of equipment and staff expenses.

In terms of their situational suitability and reliability, the use of one method over another depends on the specific attributes being measured. However, not all methods are suitable over the full range of particle sizes. Each method can be considered consistent only when being compared with results of repeated measurements,
provided the same experimental setups are used. Given the wide variety of variables involved with these techniques, care must be taken when comparing results with previous analyses outlined in research publications. Provided the authors have explicitly stated their assumptions and implementation of the technique, there should be no issue. However, in some cases, these have been omitted.

Extensive research has been conducted in this field in recent years; however, there is a need for standardisation of the measurement techniques to allow for comparisons to be made, particularly with laser diffraction based systems.

It is evident from the studies reviewed that there is a large variation in the way that particle size analysis results are presented and compared. The PSD of a wastewater sample is spread over a number of orders of magnitude; therefore, it may not always be necessary (or possible) to measure the full distribution. Some performance metrics (SS, COD/BOD and turbidity) are affected more by particular size ranges.

2.5.1 Conclusions

Summarising some of the key findings relating PSD to established performance metrics:

- TSS is more significantly affected by larger particles than smaller particles.
- Turbidity is caused by large numbers of smaller particles (colloids).
• There is a tendency for larger, more compact particles to be formed at higher DO concentrations.

• PSD is an important metric for evaluation of the BOD removal process.

From the evidence presented in this review, it can be concluded that PSD yields valuable information regarding the performance of a wastewater treatment plant and potentially represents a reliable, cost-effective method for monitoring and control of wastewater treatment plants.

From the review of the methods for determining PSD, an opportunity can be seen for development of automated systems that focus on the lower end of the PSD scale i.e. similar to particles that cause turbidity, and systems that look at larger sludge monitoring. Using the relationships established by previous literature, inferences can be made about other quality metrics from turbidity and this can then provide real-time performance information relatively quickly and more regularly than current methods. However, the results of some of these measurements are not suitable as surrogate measurements of other parameters.

Similarly, the lack of standardisation and conformance to Standard Methods in practice, has highlighted the need for standardised systems of measuring SSV. Such a system could provide more regular monitoring of sludge characteristics in unmanned WWTPs where there is little monitoring outside of that required for compliance reporting.

The next chapter details the development of an automated system for determining
SSV of mixed liquor samples that adheres to Standard Methods.
Chapter 3

Image processing techniques for determination of Settled Sludge Volume

3.1 Introduction

Determination of the SVI is key to describing the settling characteristics of sludge in the aeration process of WWTPs. The two core components required for SVI calculation are SSV and SS.

While the measurement procedure for SSV is generally defined by national or international standards, in practice, a wide variety of vessel sizes and shapes are used by operators to monitor WWTP performance. Furthermore, differences in how
these tests are carried out can lead to poor quality data, inefficient WWTP operation and a lack of comparable metrics for WWTP operational monitoring. There is a compelling need to improve operational performance of WWTPs to meet the increasingly stringent legislation regarding discharge limits.

A range of different vessel volumes, shapes and sizes are utilised e.g. settlometers and Imhoff cones leading to difficulties in comparison of results between WWTPs. Even where compliance monitoring is performed as per regulatory requirements there is a need for improved automated systems to perform monitoring between the required reporting deadlines [158]. Automated monitoring systems would allow for more refined process control, rather than operating a WWTP for the worst-case scenario all of the time as is often the case [5].

The aim of this study was to utilise a novel image-processing system (AutoSSV) to (i) determine its efficacy in describing SSV and (ii) compare different methodologies for measurement of SSV. The AutoSSV system was tested using samples from various WWTPs and the results compared to those determined by standard manual measurement. Both standard and modified settlement tests were conducted on 30 mixed liquor samples, with modified settlement tests consistently resulting in lower SSV measurements. Results from the study showed a strong correlation between the SSV measurements provided by the AutoSSV system and results obtained from current manual measurement methods.

The proposed technique would help to standardise the measurement in practice
and increase the frequency of monitoring, particularly in small scale remote WWTPs where there may not be permanent operators on site, to perform manual performance monitoring for efficient and effective operation.

The system proposed in this chapter was published in Water, Science and Technology in 2018 [159].

3.1.1 Proposed Solution

Previous studies have proposed the use of cameras and image processing techniques as a means of measuring the SSV30 of mixed liquor sludge [160,161]. Potential drawbacks of the previous systems include: (i) A low resolution camera (390x230) with 230 pixels along the axis of interest; (ii) the system was not capable of performing modified SSV measurements, such as the stirred SSV; (iii) no discussion was presented regarding the effects of the floating scum layer formed at the air-water interface, that can affect automated sludge height estimations. This study presents a system suitable for autonomous determination of the stirred SSV of mixed liquor wastewater samples from aeration tanks. The system images mixed liquor in a transparent vessel, and from these images, calculates not only the SSV30 measurement with stirred settling, but also the rate of settling. This is achieved using a high-resolution digital camera and image processing techniques that provide a repeatable and objective measurement that could be deployed as an autonomous remote monitoring solution; particularly in remote WWTP locations, where currently minimal
monitoring is performed.

The vertical settling velocity of particles in a liquid over a period is captured via a sequence of time stamped images. From these images a colour intensity profile is extracted to measure the settling sludge height as a function of time. Results are presented for a dataset of 30 mixed liquor samples collected over several months from a variety of activated sludge municipal WWTPs. The AutoSSV is then correlated with the manually measured equivalent. In addition to the SSV value defined at 30 minutes, the rate of settling for the sludge can also be examined at a set time interval within that 30 minutes to find intermediate rates of settling. This is achieved utilising inexpensive hardware, specific lighting conditions and standardised vessels. Such a system could potentially replace the need for manual measurement of SSV and simplify the SVI measurement process.

3.2 Measurements and Methods

3.2.1 Sample Collection

Mixed liquor samples from 3 Irish municipal WWTPs, referred to as: TP01, TP02 and TP03, were collected over a 7-month period. Each WWTP received influent from a combined storm and sewer line in their municipal area. The WWTPs were selected for this case study as they provided a range of sludge types with a broad spread of SSV30 values (based on historical data). Stirred and quiescent settlement tests were
carried out on samples of mixed liquor collected from each site. The WWTPs from which these samples were collected are all publicly operated municipal activated sludge systems of various size, and are listed in Table 3.1.

Two of the WWTPs sampled had two completely mixed aeration tanks operating in parallel (and thus independently from each other) and in those cases (TP01 and TP02) samples were collected from both tanks, identified and tested. The letters 'a' and 'b' following the Plant Reference are used to identify separate tanks at a facility. The samples were collected directly from the final aeration tanks immediately upstream of the outlet to the secondary clarifier system using a manual grab sampler (Wheaton Science Products). The WWTPs from which samples were collected to carry out laboratory testing are also listed in Table 3.1. A minimum volume of 25 L of mixed liquor was taken at each collection event in the same manner as used by the on-site operators. The collected samples were transported to the laboratory for immediate testing. Prior to use or decanting to a smaller vessel, the sample storage container was inverted a number of times to ensure homogeneity of the sample and to negate the effects of any settlement that might have occurred during transport or storage.

3.2.2 Control Results

Quiescent settlement tests were carried out by filling the settlement vessel [48] to the required volume with a well-mixed sample of mixed liquor. The height of
Chapter 3: Image processing techniques for determination of Settled Sludge Volume

Table 3.1: Sampled WWTPs Settlement Tests

<table>
<thead>
<tr>
<th>Ref</th>
<th>Samples</th>
<th>Design Size (Population Equivalent)</th>
</tr>
</thead>
<tbody>
<tr>
<td>TP01a</td>
<td>8</td>
<td>25000</td>
</tr>
<tr>
<td>TP01b</td>
<td>15</td>
<td>25000</td>
</tr>
<tr>
<td>TP02a</td>
<td>2</td>
<td>6000</td>
</tr>
<tr>
<td>TP02b</td>
<td>4</td>
<td>6000</td>
</tr>
<tr>
<td>TP03</td>
<td>1</td>
<td>4000</td>
</tr>
</tbody>
</table>

the sludge-supernatant interface was recorded at 1-minute intervals for the first five minutes and at five-minute intervals thereafter for the remainder of test duration.

Stirred settlement tests were carried out in a similar way by filling the settlement vessel to the required volume with a well-mixed sample of mixed liquor and activating the stirring device (a stainless-steel two-rod assembly suspended over the vessel) as specified in Standard Methods 2710 C. The height of the sludge supernatant interface was also recorded at one-minute intervals for the first five minutes and at five minutes intervals thereafter for the remainder of the test duration.

Vessel Specification

The apparatus requirements for Standard Methods SSV 2710 C is described as a 1 L graduated cylinder equipped with a stirring mechanism consisting of one or more thin rods rotating at no more than 4 rpm [48]. A polycarbonate settleometer vessel (USA Blue Book 1.3 L Settleometer Kit) [162], shown in Figure 3.1, of capacity 1.3
L but graduated to 1 L was used in the laboratory work to replicate the Standard Methods SSV 2710 C test vessel. While curved vessels can present problems with light reflection when imaged (as evident in Figure 3.2), it was decided to continue using them as they provide a more comparable analogue to current measurement methods, since both vessel diameter and height are factors that affect the results of settling tests [163, 164]. Settling in a flat faced vessel is difficult to interpret due to the complex fluid dynamics in the corners of the vessel.

Although the correct vessel size and shape is outlined in the Standard Methods, as previously mentioned, in practice this is not always used by plant operators [38]. One additional goal of the proposed system was to enable standard practices on all sites (including unmanned sites), thus allowing for more comparable results between WWTPs.
3.2.3 Imaging System

An automated laboratory rig was constructed to carry out SSV and stirred SSV testing. The unit comprised two independent aluminium frames, each housing two standard settleometer vessels, complete with two motors and gearboxes for stirring. A two-rod stirring assembly was constructed to provide stirring action in the 1.3 L vessel. The settleometer vessels were mounted on laboratory jacks, which were integral to the frame, and whose platforms could be raised and lowered manually to allow installation and removal of the settlement vessels and stirrer assemblies (Figure 3.3). The rig allowed for simultaneous testing of two samples with variable stirring speed. Lighting was provided via a series of high intensity LED devices. Light
placement was configured to provide optimum contrast between the background and the sludge column for layer profiling.

A calibrated scale at the front and back housing of the settleometer rig allowed the centre of the camera axis to be determined via parallax methods, reducing the need for precise alignment of the system.

The four motors were powered by independent variable DC power supplies. By varying the voltage, the motor speed could be changed. A stainless steel two-rod stirring assembly, capable of rotating at no more than 4 rpm, as per the requirements of Method 2710 I [48] was constructed.

### 3.2.4 Camera

For this image acquisition, a commercially available DSLR Camera (Canon EOS 600D) was selected. It featured an 18 MPixel CMOS image sensor, with an F number range of 5.6-36 and an ISO range of 100-6400. The camera was controlled using external circuitry to both automate the acquisition and to avoid any interference with the camera alignment and focus. This remote shutter control was triggered by the stirring mechanism attached to the motor assembly of the test rig. The motor speed in RPM was used to determine the number of photos taken per minute, with one photo taken for every revolution. The trigger circuitry emulated the button press performed by a standard external remote trigger. A small plate, attached to the motor shaft, passed through a light gate on every revolution and a transistor
Chapter 3: Image processing techniques for determination of Settled Sludge Volume

Figure 3.3: Aluminium test rig for conducting SSV tests. Included adjustable jack stands, stirring mechanism and settlement vessel. Constructed rig shown in Figure 3.4.
Figure 3.4: Test rig as built from schematic shown in Figure 3.3. The frame is constructed from aluminium extrusions.
acted as a switch. The stirrer was constructed from stainless steel rods and was attached to the keyed shaft of the motor via a set screw. This allowed the camera to be triggered when the spindle/paddle was in the correct orientation in order to not interfere with the level estimation.

To ensure a repeatable imaging geometry for the camera, another aluminium structure was constructed on which to mount the camera. The camera was fixed in the portrait orientation to best utilise the size and shape of the imaging sensor relative to the vessel. Additional features, such as automatic light balance and gamma correction, normally enabled by default in the camera were turned off to ensure the camera did not try to compensate for changes in light balance as the sample settled. The camera was focused on the front face of the vessel on the focal plane. Images were saved as 3456x5184 pixel JPEGs and each AutoSSV test generated approximately 750MB of data, including calibration images.

Lighting

The light source comprised several strips of broadband white LEDs placed on the two vertical uprights between the vessel and the camera, as well as another strip on the upper horizontal strut between the two. The LEDs were attached directly to the aluminium structure to aid heat dissipation. Power supplies and control systems were placed in a weather proof box. Due to the shape of the vessel, there was a tendency for reflections to originate from the centreline of the settleometer. To reduce these
reflections, all other light sources in the area were turned off during data recording. The LED placement ensured a uniform illumination of the front face of the vessel and minimised the amount of reflections on the curved face. In the data analysis/image processing stage, areas with reflections were avoided. A card was placed behind the settleometer to reduce light scatter from behind the settleometer.

### 3.2.5 Test Procedure

The test procedure utilised for data collection was as follows: The test vessel was filled with a mixed liquor sample and placed into the test rig, using the alignment pins on the platform to ensure correct position. The stirring paddle was then placed into the vessel and attached to the motor shaft (the paddle and motor shaft were keyed to ensure correct alignment.) The platform was then raised such that the 500 mL marker was in line with the centreline of the camera lens. The camera was then manually focused so that the fiducial markers at the front of the test rig were in focus; thus locating the front of the vessel on the focal plane. A similar test was conducted in parallel using the same mixed liquor sample in a non-stirred SSV test.

For volume calibration (conversion of pixel heights to equivalent volume on settleometer), the vessel was imaged once with graduations facing towards the imaging sensor. The graduations were then rotated out of the field of view so that they didn’t interfere with the subsequent image processing. Prior to commencing the 30 min SSV test, the sample was manually agitated to ensure homogeneity. The stirring
motors were then switched on and the test commenced.

For control results, a manual measurement of the sludge volume in mL was taken every minute for the first 5 minutes, and every 5 minutes thereafter. At the conclusion of the test, the images and the manual measurements were stored for processing at a later stage.

After the SSV estimation process was complete, the platform was lowered, and the stirring paddle removed. The vessels were then removed from the test rig, the sludge samples were disposed of and the vessels were thoroughly cleaned for the next test. The test procedure was repeated for 30 mixed liquor samples, collected over a 7-month period.

3.3 Processing the Results

The central algorithm developed for this system employed a multilevel adaptive threshold of the captured images to estimate sample layers, corrected for parallax effects on the volume estimates. The pixel values of vertical line profiles through the centreline of the settleometer were used to assess the settlement of the sludge column. The algorithm was also used to detect the scum layer that often formed on top of the sample as shown in Figure 3.5. Each measurement was obtained by determining the high to low transition in light intensity associated with the transition between the air at the top of liquid meniscus and the sludge water mixture, then the low to high transition as the profile transitioned to the main volume of liquid.
3.3.1 Calibration

Use was made of fiducial and graduation markers on the flask to perform real world coordinate calibration and image de-rotation correction as required. The reference image (image with graduations facing towards the image sensor) was loaded and several points of reference were found from which to measure. These are shown in Table 3.2. The centreline of the vessel in the Y direction was found by drawing a line between the front and back fiducial markers on the test rig structure and choosing the height at which there was no parallax error. The X direction was considered as the vertical centreline of the vessel. A configuration file was then written to store
these parameters for each AutoSSV test. There were minor alignment differences in subsequent tests as there are a number of moveable parts involved in the image acquisition rig. The final settled sludge height and settling rate was then determined for each sample collected.

<table>
<thead>
<tr>
<th>Reference Parameter</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Angle</td>
<td>Angle rotated to ensure vertical sections through vessel are parallel with sidewall.</td>
</tr>
<tr>
<td>X center</td>
<td>Centre Point of vessel volume in X direction.</td>
</tr>
<tr>
<td>Y center</td>
<td>Centre Point of vessel volume in Y direction.</td>
</tr>
<tr>
<td>Y 1000ml</td>
<td>Height of 1000 mL marker in Y axis.</td>
</tr>
<tr>
<td>Y 100ml</td>
<td>Height of 100 mL marker in Y axis.</td>
</tr>
</tbody>
</table>

### 3.3.2 Image Processing Algorithm

**HSV colourspace**

Each image captured by the camera was converted from the RGB (red, green, blue) colour space to the HSV (hue, saturation, value) space, and the V (value) channel extracted. The V channel provided the best contrast between the dark sludge and clear supernatant. From the initial calibration, performed on the first image, each image was rotated such that the vessel walls appeared vertical.
Extract line profile

The algorithm plotted the intensity profile and marked the level estimates as shown in Figure 3.6. An intensity profile was extracted for a column of pixels along the axis of the vessel, through the volume starting at 0 mL and running to a maximum level at the top of the sample. Due to the curved face of the vessel, a single column was chosen, rather than a wider region of interest (ROI). Due to parallax, the top of the settling sludge does not appear as a horizontal line, therefore averaging across the width of the vessel was not possible. The single column intensity profiles were found to be inconsistent due to the inhomogeneity of the sludge caused by the floc composition. Extracting smooth profiles is a challenge with this type of imaging and in general, median based filters are effective at removing impulse noise in digital images with small signal distortion [165].

Differentiate the line profile

To find the sludge-supernatant interface, the turning points of the intensity profile were found from the derivative of the profile. As observed in Figure 3.6 additional negative peaks can be seen at the air-water interface. To combat this, only transition regions where the signal derivative dropped below the threshold \( T \) (empirically derived as 0.1 times the minimum signal derivative) were logged. The value of 0.1 was chosen as it indicates the significant peaks in the profile derivative that are sudden drops in intensity along the profile. The turning point locations were used
to estimate the liquid meniscus and settled sludge layer.

**Finding the turning points**

A state machine was used to find the coordinates of the turning points. Each point \( X(i) \) of the profile was compared to the threshold \( T \) and if \( X(i) < T \) the state output was **True**, otherwise the state output was **False**. The truth table for the state machine is shown in Table 3.3.

Table 3.3: Truth Table for State Machine that locates turning points intensity profile.

<table>
<thead>
<tr>
<th>( X(i) )</th>
<th>( X(i-1) )</th>
<th>State</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>0</td>
<td>Out of Region</td>
</tr>
<tr>
<td>0</td>
<td>1</td>
<td>Entering Region</td>
</tr>
<tr>
<td>1</td>
<td>1</td>
<td>In Region</td>
</tr>
<tr>
<td>1</td>
<td>0</td>
<td>Leaving Region</td>
</tr>
</tbody>
</table>

The processes in each state are outlined as follows:

- **Entering Region**: Start point was defined as index of \( X(i) \).
- **In Region**: No change, proceeded to \( X(i+1) \).
- **Leaving Region**: End point was defined as index of \( X(i) \)
- **None**: No change, proceeded to \( X(i+1) \).

From the output of this state table, the indices of the negative peaks of the derivative of the section profile were identified, thus locating the supernatant-sludge
Figure 3.6: Image of partial settlement after 3 minutes. Graduate scale overlaid on vessel, with 100 mL divisions. Sludge height also marked. Plot of vertical section through mixed liquor sample and its derivative shown below it. Sludge water interface marked on both plots.
interface. In situations where multiple peaks satisfy the threshold conditions, the largest index value is chosen to be the peak of interest.

**Convert to sludge height**

A plot of the section profile was generated for each image with the sludge height marked, as shown in Figure 3.6. For each image, the sludge level was computed in mL, timestamped and logged. From this log of settlement levels, a settlement curve was generated. The final image (taken 30 minutes from the start of the test) provided the SSV30 measurement.

Collection of mixed liquor samples from WWTPs was the major source of labour associated with this experimental procedure. The image collection time was dependent on the settling test i.e. 30 minutes. Data processing time was minimal with this system; an entire image set could be processed by computer in 20 minutes. The processing time could be overlapped with the settlement test, such that each image is processed as it is taken, thus reducing the total time required and implementing real-time operation.
Figure 3.7: Block Diagram of the testing methods. Blocks inside the shaded box are repeated for each image taken. Blocks outside are repeated for each sample collected.
3.4 Results

3.4.1 Statistical Analysis of Manual vs AutoSSV

The raw results from the image processing were expressed as a sludge height in pixels. This value was then scaled to a value for sludge volume in mL using the 100 mL and 1000 mL marker points, defined in the configuration file, as points of reference for calibration.

The results of this analysis are presented in Figure 3.8, with the full table of results included in Table 3.4. A high level of correlation was found between the two measurement techniques for the SSV30 measurement. Pearson’s R showed an average $R = 0.99$ with a p-value $< 0.01$, suggesting the results are significant. A strong linear correlation (Pearson’s R) does not always indicate agreement between two datasets; as an additional check, Lin’s Concordance Correlation Coefficient (CCC) (a measure of agreement for variables with continuous data that compares the distance between the line of best fit of the datasets [166]) was also computed. Using Lin’s CCC an average correlation of 0.99 was observed.

A two-sample T-Test yielded a P-value $< 0.9$ in favour of the null hypothesis that the results of the automated and manual SSV were equivalent. There was an average error of 3.7% across the entire dataset of SSV30 values between the manual and the automated measurements (using the manual measurement as the standard).
Table 3.4: SSV30 Results for 30 Mixed Liquor samples from 3 Municipal WWTPs, some containing multiple aeration tanks

<table>
<thead>
<tr>
<th>Sample</th>
<th>Test Date</th>
<th>Plant Ref.</th>
<th>SSV30 Manual (ml)</th>
<th>SSV30 Auto (ml)</th>
<th>Absolute Difference (ml)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>18/02/2014</td>
<td>TP01a</td>
<td>420</td>
<td>429</td>
<td>9</td>
</tr>
<tr>
<td>2</td>
<td>18/02/2014</td>
<td>TP01b</td>
<td>450</td>
<td>450</td>
<td>0</td>
</tr>
<tr>
<td>3</td>
<td>21/02/2014</td>
<td>TP01b</td>
<td>320</td>
<td>333</td>
<td>13</td>
</tr>
<tr>
<td>4</td>
<td>24/02/2014</td>
<td>TP01a</td>
<td>220</td>
<td>220</td>
<td>0</td>
</tr>
<tr>
<td>5</td>
<td>25/02/2014</td>
<td>TP01b</td>
<td>190</td>
<td>177</td>
<td>13</td>
</tr>
<tr>
<td>6</td>
<td>26/02/2014</td>
<td>TP01a</td>
<td>150</td>
<td>175</td>
<td>25</td>
</tr>
<tr>
<td>7</td>
<td>27/02/2014</td>
<td>TP01b</td>
<td>610</td>
<td>611</td>
<td>1</td>
</tr>
<tr>
<td>8</td>
<td>03/03/2014</td>
<td>TP01a</td>
<td>350</td>
<td>357</td>
<td>7</td>
</tr>
<tr>
<td>9</td>
<td>03/03/2014</td>
<td>TP01b</td>
<td>490</td>
<td>481</td>
<td>9</td>
</tr>
<tr>
<td>10</td>
<td>05/03/2014</td>
<td>TP01a</td>
<td>330</td>
<td>325</td>
<td>5</td>
</tr>
<tr>
<td>11</td>
<td>10/03/2014</td>
<td>TP01a</td>
<td>780</td>
<td>772</td>
<td>8</td>
</tr>
<tr>
<td>12</td>
<td>10/03/2014</td>
<td>TP01b</td>
<td>490</td>
<td>495</td>
<td>5</td>
</tr>
<tr>
<td>13</td>
<td>12/03/2014</td>
<td>TP03</td>
<td>240</td>
<td>246</td>
<td>6</td>
</tr>
<tr>
<td>14</td>
<td>18/03/2014</td>
<td>TP02a</td>
<td>620</td>
<td>609</td>
<td>11</td>
</tr>
<tr>
<td>15</td>
<td>18/03/2014</td>
<td>TP02b</td>
<td>270</td>
<td>289</td>
<td>19</td>
</tr>
<tr>
<td>16</td>
<td>19/03/2014</td>
<td>TP01a</td>
<td>640</td>
<td>607</td>
<td>33</td>
</tr>
<tr>
<td>17</td>
<td>19/03/2014</td>
<td>TP01b</td>
<td>600</td>
<td>593</td>
<td>7</td>
</tr>
<tr>
<td>18</td>
<td>19/03/2014</td>
<td>TP03</td>
<td>290</td>
<td>288</td>
<td>2</td>
</tr>
<tr>
<td>19</td>
<td>19/03/2014</td>
<td>TP02a</td>
<td>980</td>
<td>942</td>
<td>38</td>
</tr>
<tr>
<td>20</td>
<td>19/03/2014</td>
<td>TP02b</td>
<td>390</td>
<td>392</td>
<td>2</td>
</tr>
<tr>
<td>21</td>
<td>07/08/2013</td>
<td>TP02b</td>
<td>400</td>
<td>362</td>
<td>38</td>
</tr>
<tr>
<td>22</td>
<td>28/08/2013</td>
<td>TP01a</td>
<td>550</td>
<td>628</td>
<td>78</td>
</tr>
<tr>
<td>23</td>
<td>02/10/2013</td>
<td>TP02b</td>
<td>300</td>
<td>279</td>
<td>21</td>
</tr>
<tr>
<td>24</td>
<td>20/01/2014</td>
<td>TP01b</td>
<td>420</td>
<td>438</td>
<td>18</td>
</tr>
<tr>
<td>25</td>
<td>21/01/2014</td>
<td>TP01b</td>
<td>420</td>
<td>423</td>
<td>3</td>
</tr>
<tr>
<td>26</td>
<td>21/01/2014</td>
<td>TP01b</td>
<td>480</td>
<td>468</td>
<td>12</td>
</tr>
<tr>
<td>27</td>
<td>21/01/2014</td>
<td>TP01b</td>
<td>410</td>
<td>408</td>
<td>2</td>
</tr>
<tr>
<td>28</td>
<td>21/01/2014</td>
<td>TP01b</td>
<td>450</td>
<td>453</td>
<td>3</td>
</tr>
<tr>
<td>29</td>
<td>23/01/2014</td>
<td>TP01b</td>
<td>440</td>
<td>428</td>
<td>12</td>
</tr>
<tr>
<td>30</td>
<td>24/01/2014</td>
<td>TP01b</td>
<td>330</td>
<td>300</td>
<td>30</td>
</tr>
</tbody>
</table>
Figure 3.8: Scatter of SSV30 results, Manual vs Auto, with line of best fit.

\[ R^2 = 0.99 \]
3.4.2 Stirred vs Unstirred settlement tests

A comparative study of stirred and unstirred settlement was undertaken on 30 mixed liquor samples to highlight the difference in SSV30, the results of which are shown in Figure 3.9. A clear difference can be seen between the two tests, with the stirred method generally resulting in a lower SSV30 measurement than the unstirred. A T-test yielded a p-value of 0.006, suggesting a significant difference. Stirred settlement is generally considered more representative of the conditions in the settling tanks in WWTPs, as it was introduced to the Standard Methods 2710 C for determining SSV in 1980 [48]. Correlating the stirred and unstirred methods resulted in a $R = 0.83$, with unstirred settlement tests generally resulting in a higher SSV measurement after 30 minutes than the stirred test on the same sample. From these results, the benefit of an automated system for measuring stirred SSV can be seen as stirred settlement is a better indication of settling characteristics in the clarifier.

3.4.3 Sludge settling rate

To examine the sludge settlement testing system robustness, the SSV30 dataset was extended by considering the intermediate SSV height measurements as the final SSV measurement. Manual and automated system measurements were examined at select intervals in the settling process, every minute for the first 5 minutes and every 5 minutes subsequently, yielding 10 points of comparison between the two methods. The results of this analysis are shown in Figure 3.10.
Figure 3.9: Comparison of SSV30 stirred and unstirred measurements
Figure 3.10: Plots of Sludge height over time for 10 mixed liquor samples

As can be seen from the results, there was a high level of agreement between the manual and automated system. However, there were several cases where the scum layer at the top of the sample was erroneously selected by the algorithm, as the top of the settling sludge-supernatant interface. A line of best fit was applied and both Pearson’s R and Lin’s CCC computed, with the results shown for each sample in Table 3.5. Despite the outliers (seen in Figure 3.10), caused by the floating scum at the top of the sample, many of the correlation results were greater than 0.9 between both methods, with an average value of 0.94 and 0.91 for Pearson’s R and Lin’s CCC respectively. The p-value was < 0.001 indicating the results are significant. A two-sample T-Test yielded a P-value < 0.91 in favour of the null hypothesis that the results of the automated and manual stirred SSV were equivalent.
Chapter 3: Image processing techniques for determination of Settled Sludge Volume

Table 3.5: Summary of results

<table>
<thead>
<tr>
<th>Measurement</th>
<th>n</th>
<th>R</th>
<th>P-value</th>
<th>$R^2$</th>
<th>T-test P-value</th>
<th>Lin's CCC</th>
</tr>
</thead>
<tbody>
<tr>
<td>SSV30</td>
<td></td>
<td>0.99</td>
<td>&lt;0.01</td>
<td>0.985</td>
<td>0.66</td>
<td>0.99</td>
</tr>
<tr>
<td>Incremental</td>
<td></td>
<td>0.94</td>
<td>&lt;0.001</td>
<td>0.88</td>
<td>0.91</td>
<td>0.91</td>
</tr>
</tbody>
</table>

3.4.4 Discussion

There were a few challenges associated with detection of sludge height in the stirred settlement tests. As shown in Figure 3.11, the layer of floating scum at the top of the mixed liquor did, on occasion, interfere with the measurement of the settling sludge-supernatant interface. This was caused by less dense particles in the sample that gradually rose to the top as the rest of the sludge settled. Also, in two or three cases, a large amount of sludge rose to the top of the vessel part way through the settling test as shown in Figure 3.12. However, in situations where a significant amount of sludge floats to the top of the sample, the SSV30 results are unlikely to be significant as this is generally an indicator of much greater issues with the treatment process such as denitrification, high levels of fats, oils or greases in the mixed liquor or bulking sludges caused by nutrient deficiencies [167]. The system presented here could easily be used to detect such problems, by visually inspecting the images, or in future using image processing to measure the thickness of the scum layer and would be particularly effective in WWTPs without permanent on-site operators. Equally,
there are situations with high sludge content samples that effectively don’t settle after 30 min, and therefore deliver no measurable result.

The proposed system has advantages over previous image processing-based SSV measuring systems: previous systems that examined SSV extracted an intensity profile and computed the slope from the origin to each point on the profile i.e. 229 calculations per image collected and the maximum slope was assumed to be the sludge height [160]. This multiple slope-based method does not account for thick floating scum layers at the top of the vessel. Whereas, the system proposed in this chapter uses an improved adaptive threshold to narrow down the points of interest from the derivative to fewer negative peaks of interest and then chooses the most significant.
Chapter 3: Image processing techniques for determination of Settled Sludge Volume

Figure 3.12: Large floating scum layer that rose part way through settlement test.

Existing image-based systems for performance monitoring cannot examine the stirred SSV, as they were not designed to include such a stirring mechanism [161]. Figure 3.9 showed the difference in SSV30 values between stirred and un-stirred settlement tests. Stirred settling are a requirement of the Standards Methods 2710 C since 1980 and thus should be included as part of an automated SSV measurement system.

A camera-based system such as this one has advantages: it could self-diagnose its state of calibration because a clean empty vessel could be imaged and could then be used to compare the vessel clarity or cleanliness with those of previous tests and either clean it or compensate for it in the algorithms.
3.5 Conclusions

In this chapter, an automated method of determining SSV using image processing is proposed. Studies have found that compliance with SSV measurement techniques outlined in Standard Methods (or equivalent guidelines) is currently poor. Replacing the manual procedure with a system based on image processing provides both objectivity and repeatability by reducing the requirement for costly human intervention.

Results have shown that it is possible to determine SSV with a high level of accuracy using image processing. Control tests as per the accepted Standard Methods were utilised to determine the accuracy of the proposed system. There was a clear correlation between both methods, not only for the SSV30 measurements, but also for the incremental sludge height measurements taken during the settling phase.

In future, it is envisaged that turbidity of the supernatant could be examined, as it is indicative of effluent turbidity. From work conducted previously [54, 168], it is known that IR light is required for turbidity analysis, as the dissolved organic matter present absorbs broadband white light in different ways. However, the camera used for this analysis was unsuitable for viewing IR light, as like most consumer cameras, it contains an IR-UV filter.

The proposed system has the advantage of being automated if combined with a sample collection system. This could be used to provide a simple performance metric that could be remotely monitored on a more frequent basis than is currently the case.

It is clear that the energy consumption of wastewater treatment plants must be
reduced. Improvements in operational monitoring using low-cost and robust sensors can reduce this energy consumption by allowing real-time feedback and thus more efficient and responsive process management. As discharge regulations become more stringent, performance monitoring is becoming increasingly important. Implementing the proposed SSV system could reduce labour requirements, improve monitoring frequency for WWTPs and potentially reduce energy usage while increasing WWTP efficiency.
Chapter 4

Imaging and Wastewater Effluent Analysis

4.1 Introduction

This chapter focuses on an imaging system for wastewater effluent. The specific lighting and hardware utilised are described and the imaging setup is validated, using a subjective test to estimate turbidity.

Turbidity can be described as a measure of how cloudy a fluid is, and is widely used as a performance indicator in water treatment systems [80–82, 169]. It is a characteristic of a fluid, that expresses the amount of light scattered by particles in the fluid when light is shone on it. In wastewater treatment, turbidity is primarily caused by the colloidal (< 1µm) particles that dominate frequency particle size
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distributions (PSD)[83]. Turbidity is one of the simplest measurements that can be conducted on an effluent sample; however, thus far there is no single agreed standard used in practice. The Standard Methods [48] outlines the procedure for effluent testing, including measuring turbidity, but as indicated by Hannon et al., these guidelines are not always adhered to [38]. Measurements are often made subjectively rather than objectively, and that is the focus of this chapter: validating subjective turbidity measurements made by visually inspecting an effluent sample. Several commercially available turbidity measurement systems exist that use a variety of units of turbidity, the two most common units of turbidity being: Nephelometric Turbidity Units (NTU) and Formazin Attenuation Units (FAU). The units used are dependent on the light measuring sensor orientation with respect to the light source. For NTU, light scatter through the test sample is measured orthogonally to the source, while for FAU light is measured straight through the test sample, in line with the source [170].

This chapter discusses the development of a system for visually monitoring water characteristics using a camera to take an image of a fluid sample, as well as creating a database of images for extraction of visual trends over time. A case study of wastewater treatment plant effluent was undertaken to validate the system. The usefulness of these images for turbidity estimation was examined using a subjective test. The results of this test were then compared with established methods to determine turbidity. The system could also be applied to other fields of fluid analysis
that use turbidity measurements such as water treatment and marine applications [100].

### 4.1.1 Relating Turbidity to WWTP Metrics

A number of studies have related turbidity to wastewater characteristics, particularly particle size [84, 111, 171]. More recent studies are continuing this work to establish a relationship between turbidity and total suspended solids (TSS) as well as chemical oxygen demand (COD) [25, 82, 89, 172]. While an effluent stream with a low turbidity is not a definitive indication of a treatment plant working to specification, a high effluent turbidity definitively indicates that the treatment process is not operating correctly [173–176]. While turbidity cannot be used as a replacement for direct TSS measurements, an increase in turbidity generally indicates an increase in TSS [61]. While not as accurate as a direct measurement, the relationship between turbidity and TSS, as well as COD, could be utilised to improve operational decisions in WWTPs, as turbidity measurements are much less labour-intensive than either TSS or COD measurements [85].

### 4.1.2 Challenges with current effluent monitoring practices

WWTP operators currently measure turbidity both for regulatory purposes and as a plant performance metric. Turbidity is often used as a relative (rather than absolute) metric; additionally it can be used as a surrogate measurement for other fluid
characteristics. Many of the common monitoring procedures in small-scale WWTPs (particularly those in remote rural areas) are not standardised; therefore, interpretation of the results (including effluent turbidity) for plant operational control is almost entirely subjective [38]. The Water Framework Directive (WFD) [43] along with the INSPIRE Directive (Infrastructure for spatial information in Europe) [8] highlights the importance of efficient management and effective control and monitoring for reliable treatment plant performance.

### 4.2 WWTP monitoring

Determining the properties of treated water is a key aspect of performance monitoring in water and wastewater treatment systems. However, a lack of real-time information is one of the most significant barriers to effective monitoring. Regular manual testing is an effective, if inefficient means of ensuring that a system is operating within specified limits. In-line sensors are often deployed in lieu of on-site manual testing; however, these sensors require regular calibration to ensure consistent results. Therefore, sensor-based automated turbidity measurements are often bypassed in favour of manual subjective measurements, which in turn can be infrequent and costly. In some cases, sensor-based technologies are bypassed in favour of a simple visual assessment of clarity [64].

This study presents a novel method for regularly monitoring fluid turbidity remotely, without incurring the costs of travelling to the WWTP. From the images
collected by the remote monitoring system, an estimation of turbidity can be made and acted upon accordingly. Initially, the measurements are made using a subjective test quantifying effluent turbidity. The subjective test results are compared to those from a spectrophotometer, an established method of turbidity measurement. This subjective visual assessment was chosen as previous studies have highlighted the importance of visual clarity over nephelometric turbidity measurement [98, 177]. It is not necessary that visual inspection of sample images would yield exact turbidity values, but an estimate based on trends is feasible.

The proposed method provides the infrastructure required for a visual record keeping system of the fluid characteristics. This could be used alongside current reporting practices to help improve performance monitoring in a variety of applications.

4.3 Sample Handling & Imaging

4.3.1 Sample Collection and Control Results

For this case study, two different and independent datasets were examined. Firstly, 5 wastewater effluent samples were collected from 5 different municipal WWTPs. To increase the dataset size, these five were then mixed together in various proportions and also diluted to give 16 different turbidity values for analysis. These mixed effluent samples varied in turbidity from 10 FAU to 250 FAU, as measured by a spec-
trophotometer. A minimum volume of 5 L of effluent was collected at each sampling event to ensure that each sample analysed was representative of the effluent stream. The collected samples were transported to the laboratory for immediate testing. A second dataset was recorded using synthetic wastewater from a laboratory-scale dairy wastewater treatment system [178]. These samples were taken over a 3-day period and diluted two-fold (concentration reduced by half), in the case of a high turbidity, to yield more data points for analysis. The sample size for this dataset was 15 and the turbidity values ranged from 30 to 140 FAU, as measured by the spectrophotometer. Prior to use or decanting to a smaller vessel, the sample storage container was inverted a number of times to ensure homogeneity of the sample and to negate the effects of any settlement during transport or storage.

For validation of the proposed system, all results were compared to turbidity measurements made using a calibrated spectrophotometer manufactured by Hach (Model: DR2010). A spectrophotometer was chosen because the process of performing the test is robust, repeatable and the results were calibrated to a deionised water standard that is readily available [137]. The spectrophotometer was calibrated as per manufacturer’s specifications for turbidity calculations, and the wavelength of light of the spectrophotometer used in the experiments was 860 nm. Two samples were taken for each turbidity level and tests were conducted in triplicate to reduce experimental error, and to examine the standard deviation.

Effluent samples from the case-study WWTPs were imaged in a 4 L glass vessel...
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Figure 4.1: Imaging setup with vessel on left, illuminated from right hand side of vessel and the front face is imaged.

Figure 4.2: Imaging conducted under blackout material to avoid interference from light in the room. Vessel placed on magnetic stirrer for agitation of sample
as shown in Figure 4.1 (schematic view of the rig) & Figure 4.2 (photographic view of the laboratory rig). Samples were agitated by a magnetic stirrer placed in the vessel, thus ensuring the particles did not settle over the course of the test. To avoid any interference from environmental lighting or reflections, the test rig was surrounded by blackout material. For this study, orientation of the imaging sensor relative to the light source was examined and placing the camera perpendicular to the light source was found to be more suitable for two reasons: firstly, a reduction in the amount of light energy directed towards the camera allowed a wider range of exposures to be utilised without saturating the image sensor. Secondly, the geometry was such that light attenuation could be viewed as a simple function of fluid depth across the width of the image.

![Figure 4.3: Samples of increasing turbidity. Original image on left. Inverted image on right. On the simplified scale, the turbidities are approximately 1, 5 and 10 from top to bottom](image)

The resultant images were still quite dark, therefore for the test, the images were inverted (i.e. the negative of each image was used) to present a black on white
view. Initial feedback from test participants found that the negative image provided a better contrast between the particulate and the liquid. Figure 4.3 shows images of three different samples, with a range of turbidity values from 5-110 FAU, the original presented on the left and the inverted image on the right.

4.4 Camera

A 1.2 MPixel monochrome CMOS (complementary metal-oxide semiconductor) camera (a DFK23UM021 camera manufactured by The Imaging Source) was used. The camera was controlled using a MATLAB script, allowing the capture setting to be controlled through software to image each sample at multiple exposures for multiple measurements of each sample. Multiple exposures allowed for the optimal exposure to be found after data acquisition. The optimal value was determined by examining the image histogram and taking the exposure value that utilised the full dynamic range of the sensor (0-255) without saturating. Three images were taken four seconds apart, at each of these exposures. Having multiple images of the target allowed for postprocessing to utilise averaging in order to minimise the effects of artefacts, such as bubbles or a large floc, that might corrupt the results. Therefore, for each sample, 12 measurements were taken, with a total image acquisition time of one minute per sample. The raw image data was saved using a lossless TIFF (Tagged Image File Format) for further analysis. The IR/UV cut filter was removed from the CMOS camera to allow IR energy through to the imaging sensor. IR light sources
generally have much lower light intensity than white light sources, so to avail of the full dynamic range of the sensor, exposure times of 2 ms to 16 ms were utilised.

4.5 Remote Monitoring System

A laboratory-scale imaging prototype was designed to model a full-scale system capable of remotely logging images from a WWTP. The sample handling and dilutions were conducted manually. The architecture of the image acquisition system consisted of a readily available single-board computer, [179] (Raspberry Pi 3 Model B) and a prosumer camera; DFK 23UM021 for finer control over exposure times. Utilising the OpenCV image processing libraries [180], a Python script was written to capture each image from the camera and save it to a cloud-synced folder on the Raspberry Pi for later analysis.

4.5.1 Implementation

Two subsystems were implemented in the remote monitoring system: (a) the front-end that handled the image acquisition and local storage, and (b) the back-end that transmitted image files to a centralised location for further analysis. A fresh install of Raspbian (a variant of the Debian Linux-based operating system) was written to an SD card for installation on the Raspberry Pi. OpenCV was compiled and installed on the Raspberry Pi for interfacing with the webcam. A Python script was written (Python 2.7.0) to initialise a camera object and return an image file using

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the OpenCV library. The datetime library was then used to save the image locally with a timestamp. This local storage repository held the data until an external connection could be established to transmit the data to a cloud storage service. A Bash script for uploading to Dropbox was used to synchronise the images to a cloud storage folder by monitoring the results folder and uploading any files not already in the cloud folder; this would be an important feature for WWTPs in poor connectivity areas, as it allows for a backlog of results to be uploaded if necessary. A cronjob (cron is a time-based job scheduler in Unix-based operating systems) was set at a 10-minute interval to take an image and call the Bash script to upload the result. Another cronjob was used to delete the weekly collection of images to avoid filling up the limited storage space on the Raspberry Pi SD card. A bespoke data management system could be designed for a commercial implementation, similar to that outlined in [181].

4.5.2 Image Sizing, Data Usage and Storage Requirements

For test purposes, an image resolution of 1024x768 was used as it provided the best balance between the target aspect ratio and focus depth used in the image acquisition. This equates to a file size of 2.2 MB per image (uncompressed TIFF). To remotely log one such image every hour, over the course of one month would require approximately 1.3 GB of data, (a relatively small amount in monthly mobile data plans available in Ireland and the UK [182, 183]. Of course the image
could be compressed to save space and transmission costs (particularly where data communications services are limited), however some resolution would be lost.

4.6 Test Methodology

A subjective test paradigm was designed to manually estimate turbidity measurements from images of effluent. Twenty participants (unskilled in the area of wastewater quality estimation) were shown a series of wastewater effluent sample images on a computer screen and asked to rate their level of turbidity from 1-10. The purpose of this subjective test is to both validate the imaging technique and determine the validity of current on-site practices of visual estimation of turbidity. All participants in the subjective test were unfamiliar with the field of waste-water sampling and performance evaluation; this is common practice in subjective image and video quality assessments in other applications[184,185]. Unskilled participants were chosen as the test is designed to test their innate ability to distinguish subtle differences between images, not their knowledge of WWTP theory or practice. It is reasonable to assume that a cohort of experienced WWTP plant monitoring staff would yield better or (at worst) equivalent results. To ensure consistency, each of the samples of known turbidity were imaged and recorded using the prototype system, and these images were presented to the participants for evaluation. The use of images rather than asking participants to view physical samples provided a consistent representation of the effluent sample for analysis, regardless of when the subjective
test was conducted. A computer-based test also allowed for more efficient data management, compilation of results, and the ability to present the cohort with a broad range of images with different turbidities. The subjective test was conducted in a dedicated viewing space with low background illumination. Images were displayed on a 23 inch DELL S2340L monitor with a screen resolution of 1920 by 1080 pixels, at their original size of 1024 by 768 pixels. The background screen illumination was mid-grey, and the viewing distance was 70 cm.

4.6.1 Test Data

For the subjective test, a case-study using wastewater treatment plant effluent was carried out. Effluent samples were collected as outlined in Section 4.3 with a range of effluent turbidity values. All samples were imaged in a controlled environment in a laboratory. The turbidity of these samples varied from 20 to 130 FAU. To achieve more data points for comparison, portions of the effluents samples were mixed and diluted to generate a range of turbidity values not naturally available at the time the study was conducted. Turbidity measurements were taken using a calibrated spectrophotometer (Hach DR2010) at 860 nm as per Standard Methods 2130 [48]. To minimise errors and calculate the standard deviation, measurements were taken in triplicate and the results averaged.
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4.6.2 Interpreting Turbidity

The test implementation is similar to the methods used to subjectively assess image and video quality for multimedia applications [186]: Absolute Category Rating (ACR), where users rate images on a 5-point scale from “Bad” to “Excellent”. The range of values normally used to characterise turbidity (approximately 1-130 FAU) was deemed too granular for the participants to quantify. Therefore, to simplify the subjective test for the participants, the scores were scaled from 1-10, where 1 was the least turbid, and 10 the most turbid. Similar results could have been achieved by asking the subjective test participants to specify a range for the turbidity value e.g. 40-50 FAU; this however was deemed unnecessarily complex, whereas a 1-10 rating is a more intuitive rating convention for test participants. Scaling the turbidity measurements down to 1-10 equates to approximately a 10% reduction in resolution, that is of the same order as the average percentage deviation of 11.5% for the spectrophotometer results; therefore, subjective scores should be within similar error margins (not factoring in the inherent error associated with human observers). The scaling factor was found by taking the highest mean turbidity value, as measured by the spectrophotometer, in the set and equating that to 10 on the new scale, the other values were then scaled accordingly and rounded to the nearest integer, as follows:

\[ T_N = T_O / T_H \times 10 \]  

(4.1)
where $T_N$ is the new turbidity value, rounded to the nearest integer, $T_O$ is the original turbidity value, and $T_H$ is the highest turbidity value in the dataset.

### 4.6.3 Implementation

#### Training Data

A set of training (calibration) data was supplied to help inform the participants of the range of turbidity values that they would encounter during the test; this is common practice in visual subjective tests [185]. The calibration dataset was extracted from the database of images of effluent samples collected for this study and the spectrophotometer-generated turbidity measurement presented to each participant. Examples of three points on the scale (1, 5, and 10) are presented in Figure 4.3. The images were ordered in increasing levels of turbidity for the training data. The images used for training were not re-used in the actual test.

#### User Interface

The subjective tests were conducted using MATLAB to facilitate efficient data management and compilation of results. A representative dataset of 17 images was selected from the imaged samples, with a reasonably uniform spread of samples from low (5 FAU) to high (130 FAU) turbidity, avoiding duplicated turbidity values. The dataset of images was then ordered randomly and presented to each participant. The participants were asked to estimate the level of turbidity of each image to the best
of their ability, and their score for each image was recorded.

For completeness, both the original and the negative of the image were presented to each participant. Feedback from the tests indicated that some participants found the details in the negative image easier to discern. The user interface for the subjective test consisted of both the original image and the negative image, shown side by side and a dialog box displayed to the user to enter the test score. Also included was a reminder of the scoring system; 1 for the lowest turbidity, 10 for the highest as shown in Figure 4.4.

Figure 4.4: User interface for subjective test
4.7 Results

4.7.1 Subjective Scores

Each participant’s scores were examined and compared with the spectrophotometer results. Figure 4.5 shows the full set of subjective scores for each participant. The median, mean and standard deviation for the subjective test scores are plotted along with mean and standard deviation for the spectrophotometer results.

Figure 4.5: Comparison of the mean and standard deviation of scores for the subjective test and the spectrophotometer results
4.7.2 Statistical Agreement

The correlation of scaled turbidity values (1-10) with the spectrophotometer results were calculated, as shown in Figure 4.6, using a Pearson correlation. The method of least squares is a common approach to linear regression analysis. This approach minimises the sum of squares of the residuals for each equation, i.e. the difference between the observed value and the fitted value is minimised. Using linear regression, a clear correlation was found between both sets of data, with $R^2 = 0.94$. However, a strong linear correlation does not necessarily indicate agreement between two datasets; therefore Lin’s Concordance Correlation Coefficient (CCC) was also computed for each subjective test participant and the spectrophotometer results. CCC is a measure of agreement for variables with continuous data that compares the distance between the lines of best fit of each dataset using a bias correction factor and accounts for how far each datapoint deviates from the best fit using Pearson’s R [166]. A perfect value would be CCC = 1 indicating full agreement (equally CCC = -1 indicates total disagreement). Perfect agreement is an extremely unlikely occurrence between any two methods, and the strength of agreement required can vary by application. Lin’s CCC results, computed using MATLAB as per Lin’s definition [166], are shown in Figure 4.7.
Figure 4.6: XY plot comparing mean turbidity measurements between the spectrophotometer and the subjective test results for the 17 samples examined
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Figure 4.7: Lin’s Concordance Correlation Coefficients for Absolute and Relative scores for each test participant. Absolute values compare the actual correlation of results. Relative values are the comparison of order of the images.

The direct correlation between both results was computed, shown as the "Absolute" value in Figure 4.7. Subsequently, the images were ranked on levels of turbidity and compared with how each test participant ordered them. The lowest perceived turbidity would be taken as the first image and the highest taken as the last. This result is also shown in Figure 4.7 as "Relative". The majority of participants rated
the turbidity levels with a high degree of accuracy, reinforcing the hypothesis that an accurate indication of turbidity can be made by manual observation, even by those not familiar with the field of study. Only one of the participant’s results was remarkably lower than the average. As seen in the standard deviation of subjective scores, shown in Figure 4.5, it was possible that an uninformed individual could make significant errors. However, with more intensive training and calibration to the spectrophotometer results, this is a skill that can be improved. The potential for improvement of this skill was not investigated as the aim was validating the practice subjective evaluation of turbidity. Additionally, examination of the interquartile range (not shown) of subjective scores, reveals a narrower spread of scores at lower turbidity values than at higher values. From these results, it can be deduced that, for the participants of this subjective test, low turbidity values were estimated with a higher accuracy.

Previous studies in water quality have defined acceptable strengths of agreement ranging from 0.9 upwards [187]. Compared to the guideline levels of acceptable strength of agreement used in previous studies, the CCC is poor for the absolute comparison. However, given the subjective nature of the comparison, perhaps lower levels are acceptable. 50% of the tests have a CCC > 0.85 and 90% have CCC > 0.7. The strength of agreement is much improved for the relative comparison. 55% of tests have CCC > 0.9 and 80% have CCC > 0.85. These results are much closer to the acceptable limits described previously [187]. The mean CCC results
improve from 0.81 for the absolute comparison to 0.89 for the Relative comparison. This result shows that it is easier to make a relative comparison of turbidity than an objective estimation.

**Measuring the deviation of the subjective measurements.**

Examining the absolute deviation from the mean for each estimation in the subjective test, we can compare the two methodologies. The spectrophotometer results were calculated by taking the average of 3 repeated measurements. The mean, maximum and minimum errors (i.e. the difference between each measurement and the mean turbidity value), for each sample are shown in Figure 4.8. The average deviation from the mean over the 17 samples was 0.25 on the previously defined 1-10 scale. Similarly, the average deviation from the mean for the subjective test was found to be 1.3 on the same 1-10 scale (Figure 4.8).

The drinking water treatment directive specifies a minimum error of 12.5% and while the subjective deviation is roughly five times larger, it is still within the minimum error of 12.5 %. The benefits of such a remote monitoring system could outweigh this because it has the potential to be deployed remotely as a low-cost turbidity monitor. Converting the deviation results back to the original FAU scale results in an average error of approximately 13 FAU. The same deviation for the individual results of each test participant, as shown in Figure 4.9, shows the average deviation per person was also 1.3 or approximately 13 FAU, in which the best
Figure 4.8: Max, Min and Mean value of deviation of each spectrophotometer and subjective measurement from the average value from spectrophotometer, per effluent sample (per image).

Each participant had an average deviation of 3 FAU and the worst a mean error of 26 FAU. Over a limited dataset and using participants with no previous experience, this mean error level is a highly promising result. If skilled participants were used for this case study, the expected deviation would be lower. Equally, with repeated tests for training purposes, the results would improve.
Figure 4.9: Max, Min and Mean value of deviation of each subjective estimation from the average value from spectrophotometer per test participant.

4.8 Conclusion

The focus of this chapter was validating the on-site method for turbidity estimation using a subjective test of images of wastewater effluent. Results show that there is a clear correlation between the turbidity estimated by test participants and the measured results from the spectrophotometer. In a sample set of 20 tests, an average Lin’s CCC = 0.81 was found between subjective scores and average spectrophotome-
ter measurements. A higher CCC = 0.89 was found between the subjective order of the images and the spectrophotometer based order i.e. a relative measure of turbidity.

This study showed that an estimation can be made from images of the effluent; it is reasonable to assume that this method of measurement could be replaced by an image processing solution that is objective, repeatable, can be standardised and be deployed as a remote monitoring solution. In the next chapter, an image processing-based solution that could be remotely deployed to monitor turbidity by replacing the need for manual observation is proposed.
Chapter 5

Turbidity Analysis using Computer Vision

5.1 Introduction

The previous chapter presented an imaging system for wastewater effluent monitoring as well as a subjective manual estimation of turbidity from the resultant images. This chapter broadens the capabilities of the previously discussed image acquisition and storage system to incorporate an image-processing-based measurement.

While compliance monitoring is generally performed on a regular basis to report common WWTP discharge parameters, such as 5-day biochemical oxygen demand (BOD5), COD, suspended solids, ammonia, total phosphorus, orthophosphate and
total nitrogen to regulatory bodies, such monitoring tends to be infrequent (e.g., monthly) [16, 47, 64]. In this chapter, a novel technique for determining effluent turbidity is proposed, demonstrated using samples of effluent streams from case-study WWTPs. The proposed method was implemented using image processing and novel imaging techniques that show light absorption as a function of fluid depth. Such a system could be used in tandem with current measurements that are performed for compliance monitoring to meet discharge regulations imposed at a European level [7], and a centralised data management system [181] to provide operational information on a more regular timescale.

While turbidity is predominantly caused by suspended sediment such as silt, as well as organic materials such as algae, there can also be coloured dissolved organic matter (DOM) present which causes discoloration. The colour can vary depending on the source. Green, yellow and brown hues are the most common in wastewater effluents and in a small number of cases, mainly in industrial wastewater treatment, the effluent sample can appear orange in colour. In marine applications, humic stain refers to the tea colour produced from decaying organic matter underwater due to the release of tannins [99,188]. This DOM is another key area of research where analysis of the absorption and fluorescence of DOM at different wavelengths is leading to the development of new sensor technologies [189,190]. Instead of turbidity, visual clarity can be used as a metric [177].

The image processing system proposed in this chapter was published in Water,
5.2 Methods

This thesis presents a novel automated system for estimating turbidity, by using readily available digital cameras, specific lighting and imaging conditions, together with image processing techniques. The proposed system has been developed and tested with several effluent samples and the results presented. The concept for the image-processing-based turbidity estimation system builds and improves on two of the state of the art methods for turbidity measurement currently in use in the water/wastewater treatment industry. The proposed system removes the need for manual observation methods that operators often use to estimate turbidity in WWTP effluent streams as an informal assessment of performance [64]. Instead, it replaces the manual process by using inexpensive off-the-shelf digital camera technology, and implementing similar lighting conditions to that of a spectrophotometer to measure the attenuation of light through a sample medium.

5.2.1 Data Collection & Control Results

Effluent samples were collected as described in Chapter 4, Section 4.6.1 with a range of effluent turbidity values. All samples were imaged in a controlled laboratory environment. The sample turbidities varied from 20 to 130 FAU, as determined by a spectrophotometer. To generate more data points for comparison, portions of
different effluents samples were mixed and diluted to generate a range of turbidity values not readily available at the time the study was conducted. Turbidity measurements were taken using a calibrated spectrophotometer (Hach DR2010) as outlined in Chapter 4 at 860 nm as per Standard Methods 2130 [48]. To minimise errors and to determine the standard deviation, measurements were taken in triplicate.

### 5.2.2 Image Processing

All images were 1024x768 pixels in size, of which the fluid area occupied approximately 1024x550 pixels. Firstly, the region of interest (ROI) was defined to avoid the vessel structure, essentially cropping the images down to an area of fluid only. As shown in Figure 4.1, the vessel was illuminated from the right-hand-side; thus, the ROI began just at the right-hand side wall and extended for the full width inside the vessel.

The magnetic stirrer created areas of turbulent flow at the top and bottom of the vessel; therefore, to avoid these areas the upper and lower limits of the ROI were defined as follows: 150 pixels high, across the full width of the vessel, just above the centerline. The 150 pixel high ROI was chosen to determine an average light intensity profile across the vessel, which generates a smoother intensity profile. The rows of the ROI were then averaged to give a single light intensity profile as a function of fluid depth. From this profile, an estimated turbidity value can be calculated, expressed in pixels, henceforth referred to as camera estimated turbidity.
(CET). Examining this intensity profile, it was observed that the curve had a smooth, monotonic non-linear shape, the majority of it appearing exponential. Rather than applying a curve fit to the intensity profile, a simple single point comparison was found. Considering the light source as a step input to a black box system, the CET is an approximate measure of the fluid’s step response (or time constant $\tau$). CET was calculated from dark to light (left to right as shown in Figure 5.1), therefore the step response was defined as shown in Equation 5.1. The CET of this intensity profile was then determined as the depth in pixels at which the intensity reduced to 63% of its maximum intensity.

$$\tau = 1 - \frac{1}{e} = 63\% \quad (5.1)$$

To find the CET for a given intensity profile, firstly 63% of the maximum Intensity was found for each profile (Y value) and looping through the intensity profile values from low to high (left to right as shown in Figure 5.1), the CET was found as the X value where a horizontal line at this Y value intersects the intensity profile. These values were computed for the 3 images of each effluent sample taken at each exposure.

Examples of these intensity profiles and CET are shown in Figure 5.1 with the same effluent sample at four different exposures. The CET value was found to scale with exposure, provided the images were not saturated or under-exposed. Municipal effluent samples with turbidity values less than 30 FAU, were found to be under-exposed at 4ms exposure times, while images of samples with turbidity values greater
than 250 FAU were found to be saturated at 16 ms exposure times. In between this range, there was an average standard deviation of 11 pixels across all samples in the 30-250 FAU range, equating to approximately 4 FAU. The standard error from the mean across 4 exposures was 10 pixels or approximately 3 FAU. This scaling of results with exposure highlights the robustness of the system.

### 5.3 Results

Figure 5.2 shows nine different effluent samples imaged at 16ms exposure, the CET value (on the X-axis) was then taken as the new turbidity measurement (as a...
function of depth which was measured in pixels). The light intensity was observed to decay from right to left. The inverse relationship between CET in pixels and turbidity in FAU can also be seen in the results. The intensity profiles for samples with turbidity values of 32 and 72 FAU contained small discontinuities in the otherwise smooth decay of light intensity that were caused by floating matter in the effluent sample such as a floc or a bubble. These spikes in light intensity have the potential to give an erroneous CET value (usually a lower CET and subsequently a higher turbidity estimation when compared to comparable analyses using a spectrophotometer). This could present an issue if this spike in intensity coincided with the 63% threshold value used to calculate CET. However, the use of multiple images
(in this case 3) allowed the intensity profiles to be averaged, and the effect of these spikes to be greatly reduced; detection of these spikes is also an area for potential future work.

5.3.1 Conversion to FAU scale

For the proposed system to provide a turbidity measurement on the same scale (FAU) as the spectrophotometer, the CET was converted to a FAU turbidity value as follows: firstly, a turbidity value was measured using the proposed CET system for several water samples containing varying amounts of bentonite. These values were then correlated with the turbidity results from the spectrophotometer. Linear regression was used to find the line of best fit between the two data sets and the equation of this line used to convert CET in pixels to an FAU value (Figure 5.3). This conversion to FAU was first calibrated and then applied to independent bentonite samples, tested for proof of concept. This calibration step is also required for different sample types, e.g. synthetic, municipal and industrial wastewater, to present the results on the FAU scale. For best results, 5-10 samples spread across the range of expected turbidity values from the WWTP would be required for calibration.

5.3.2 Statistical Agreement

The two data sets considered in this paper: (a) municipal wastewater effluent and (b) synthetic dairy effluent, were examined independently as they were found to
have different light diffusion characteristics. The intensity of light diffused towards the camera by synthetic dairy wastewater is greater than that for real effluent taken from municipal WWTPs. The different composition and particle size distributions (PSD) of the two datasets were found to be the major reasons for the increase in light scattered by the synthetic wastewater. Smaller particles attenuate light much more effectively than larger particles [191]. This scattering characteristic was not an issue as the proposed system uses several different exposures to find the optimal contrast to maximise the dynamic range of light intensity shown in 5.1. The results were found to scale linearly with exposure, provided the images were not saturated or under-
exposed; thus 16 ms and 2.7 ms exposure times were chosen for the municipal and synthetic dairy based effluents respectively. This decision was based on maximising the dynamic range of light intensity shown in Figure 5.1.

**Municipal Wastewater Effluent**

For each municipal wastewater effluent sample, multiple images were taken, yielding 3 CET values. The CET measurements for each municipal wastewater effluent image are presented in Figure 5.4.
Figure 5.4: Individual image-processing-based turbidity measurements against spectrophotometer-based measurement, including standard deviation, for domestic wastewater effluent

As with the subjective analysis of theses images in Chapter 4, Lin’s CCC was also used to compare CET to the spectrophotometer results. The mean and standard deviation of results shown in Figure 5.5 highlight the potential inaccuracy of some of the CETs. A small number of standard deviation values of 20 FAU or more were present. These higher standard deviations tended to occur for lower turbidity values < 30 FAU. The entire municipal effluent CET results, shown in Figure 5.6, were
examined against the spectrophotometer results and a high level of correlation was found with a Pearson’s $R = 0.9$ and Lin’s $CCC = 0.86$.

Figure 5.5: Comparison of mean and standard deviation of results for domestic wastewater effluent for full dataset.
Figure 5.6: Correlation coefficients for the three image-processing-based measurements of domestic wastewater effluent

**Synthetic Dairy-based Effluent**

The second dataset of synthetic dairy-based effluent yielded similar results to the municipal wastewater effluent tests as shown in Figure 5.7. Figure 5.9 shows that a similar result was found with Pearson’s R=0.9 and Lin’s CCC = 0.89. The standard deviation of this dataset, shown in Figure 5.8, averages to 12 FAU. Two of the samples (9 and 15), are the primary contributors to this standard deviation value. These samples have a low turbidity and were both under- and over-estimated, but were found to cancel out over the three CET values. This behaviour was also seen in
spectrophotometer measurements. Without these samples, the standard deviation is 8 FAU.

Figure 5.7: Individual image-processing-based turbidity measurements against spectrophotometer-based measurement, including standard deviation, for dairy wastewater effluent
Figure 5.8: Comparison of mean and standard deviation of results for dairy wastewater effluent for full dataset
Figure 5.9: Correlation coefficients for the three image processing-based measurements for full dataset of dairy wastewater effluent

Table 5.1 shows that a clear correlation exists between CET and the spectrophotometer for the two datasets examined. The p-values suggest that all the results are statistically significant. There was still potential for a few outliers to be present in both datasets. However, these were localised measurements approximating turbidity from a small volume of the effluent sample. The full imaged volume may not have been entirely homogeneous. The use of repeated measurements over time has been found to reduce this error over time. Also, as mentioned previously, the spectrophotometer can have an error of 10% or more and is a single-point measurement. An imaging system has the advantage of being able to apply a multipoint analysis to
Table 5.1: Summary of correlation results for CET and Spectrophotometer.

<table>
<thead>
<tr>
<th>Dataset</th>
<th>Size</th>
<th>Pearson R</th>
<th>p-value</th>
<th>Lin’s CCC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Municipal Effluent</td>
<td>16</td>
<td>0.9</td>
<td>&lt;0.001</td>
<td>0.86</td>
</tr>
<tr>
<td>Synthetic Dairy Effluent</td>
<td>15</td>
<td>0.9</td>
<td>&lt;0.001</td>
<td>0.89</td>
</tr>
</tbody>
</table>

the measurement; therefore, three successive CET values were taken from a much larger effluent volume (spectrophotometer uses 20 ml samples, imaging system uses 2 L samples). In a commercial system examining the turbidity of a WWTP effluent stream, an operational decision would generally not be made based on one poor measurement. The trend would be monitored over time and appropriate measures taken after a series of tests confirmed a degradation in performance. Implementing such a system would be beneficial for small to medium-scale WWTPs without regular maintenance staff, as well as in large scale plants, for out of hours monitoring. It could provide an early detection system for poor performance in the treatment process and keep a record of the effluent turbidity history for study. The inclusion of image recording allows the plant operator to visually examine the sample, providing an additional level of validation of the results.

5.4 Conclusions

In this chapter, a new method of estimating turbidity using image processing is proposed.
Replacing the manual procedure with an objective system based on image processing provides both objectiveness and repeatability. By reducing the requirement for human intervention, the cost of monitoring can also be reduced. Test results have shown that it is possible to relate the diffusion of light by a turbid fluid to the turbidity of that fluid. Control tests were used to determine the accuracy of the proposed system. The spectrophotometer range of error was found to be up to 10%; though using repeated measurements can minimise this error. There was a clear correlation between the two turbidity measurement methods (spectrophotometer and CET) for the two datasets examined. The proposed system also has the advantage of being easily repeatable and could provide a simple one-dimensional performance metric that could be remotely monitored. The data recording and analysis could be performed on-site with relatively low-cost hardware. A threshold value could be defined for a turbidity level, above which a notification could be sent to alert the plant caretaker that there is an issue with the treatment system. In regions with better connectivity, more detailed data could be sent from the plant. This would allow for more regular monitoring, and saving the images for record keeping purposes or further analysis. Implementing such a system would lead to a reduction in manpower requirements and improve wastewater monitoring frequency, which is increasingly important in order to meet more stringent discharge regulations. Additionally, increased monitoring frequency could also reduce energy usage and increased WWTP efficiency. While calibration was required for the system to produce a comparable turbidity value (in
FAU), the CET value expressed in pixels is still a turbidity measurement and can be used as a metric in itself.

The next chapter concludes the findings of this thesis and discusses the future work required to combine the monitoring elements proposed in Chapters 3, 4 and 5 into one system for performance monitoring in WWTPs.
Chapter 6

Conclusions and Future Work

6.1 Project Summary

This thesis has presented a number of contributions in the area of wastewater treatment monitoring, including the development, testing and application of a camera-based monitoring system. The research was motivated by the irregular and infrequent monitoring practices, primarily caused by lack of staff, in WWTPs in Ireland. Providing a system with remote monitoring capabilities could significantly increase the frequency of monitoring, without additional staff and logistical costs, and therefore improve WWTP operational efficiency while also reducing overall energy consumption. A comprehensive literature review of physical performance metrics for wastewater treatment was presented. Current standard practices for performance monitoring were also presented. Particular emphasis was placed on visually assessed
metrics such as SSV and turbidity and how these relate to the PSD of wastewater. A review of techniques for determining the PSD of fluids was presented, describing the relative merits and demerits of each and from this review, an opportunity for development in the field was identified. Based on this review it was concluded that there was a need for systems that could automatically determine SSV and monitor effluent quality. To achieve this goal, two distinct imaging systems were designed using consumer-grade cameras (i) to conduct a SSV settlement test on mixed liquor and (ii) to measure turbidity of WWTP effluent. Image processing algorithms were designed to automate the analysis of the resultant images.

An automated imaging system for stirred SSV testing has been proposed. Settlement test were conducted on 30 mixed liquor samples from municipal WWTPs as per guidelines from the Standard Methods. The system takes regular images of the settling sludge throughout the 30-minute test. Subsequently, an adaptive thresholding algorithm is applied to each image to extract the sludge height as a function of time. From this thresholding algorithm, both the SSV30 measurement and the sludge settling rate can be found. An average correlation of $R = 0.99$ was found between the automated and manual SSV measurements. Additionally, the incremental sludge height was examined to find the sludge settling rate, and an average correlation of $R = 0.94$ was found between the manual and automated measurements.

The second system examined the WWTP effluent turbidity. Effluent samples were collected from municipal WWTPs and a lab-scale industrial treatment plant.
These samples were then imaged under IR lighting conditions such that the decay of light intensity through the medium could be seen as a function of fluid depth. The imaging technique was validated in two ways. Firstly in Chapter 4, a subjective test was designed to assess the turbidity of the effluent samples imaged. From the literature review, it was found that in Ireland, in practice, turbidity of treated effluent is assessed in the field without the use of objective measurements. This study aimed to both validate this on-site practice of visual estimation of turbidity and also validate the imaging technique proposed. The results from this study showed a clear correlation between the perceived image turbidity and an objective measurement made using a spectrophotometer. While this is not envisaged to be a practical measurement, it forms the basis for one, as well as providing a validation tool for objective measurement techniques. Comparing the subjective measure of turbidity to the established standards yielded a Lin’s CCC = 0.81, a significant result for a purely subjective analysis. Improving on this measurement with an objective, repeatable solution achieves an average CCC = 0.89.

Secondly, an objective image processing based methodology for measuring turbidity was proposed in Chapter 5. This process measures the decay of light intensity through an increasing depth of fluid. The results from this were then compared with those from the spectrophotometer and once again, a high level of correlation was found between the two techniques. This analysis was conducted on two independent datasets from different types of wastewater treatment facilities; municipal and
industrial. These datasets had differing diffusion characteristics, but with separate independent calibration, a high correlation was found.

6.2 Primary Contributions

The main contributions to this work/field are as follows:

- A comprehensive review of current monitoring practices of settled sludge volume and effluent turbidity for wastewater treatment plants in Ireland. This review focused on PSD measurement technologies used in both research and in practice for wastewater quality measurements and discussed how the PSD is related to other parameters such as sludge settleability and turbidity.

- A novel methodology for determining the stirred SSV using image processing and an adaptive thresholding technique to determine SSV30 as well as sludge settling rate.

- An imaging method for wastewater effluent was developed to capture images with turbidity information. This was extended into a remote monitoring imaging system using a consumer-grade single board computer. This imaging method allows for turbidity to be seen as a function of increasing fluid depth, from which turbidity can be estimated either manually or using image processing.
• A subjective analysis of turbid images was undertaken to assess current on-site methods of manual estimation of turbidity. Twenty participants rated the turbidity of these images and there was a high correlation between these results and those from a spectrophotometer. This test validates the current on-site method of manual turbidity estimation.

• An image processing algorithm was developed to objectively measure turbidity from images of turbid effluent. The algorithm measures the decay of IR light across an increasing depth of fluid. The decay of light was found to correlate with the turbidity measurements from a spectrophotometer. This is an important development for effluent monitoring in the wastewater treatment field, where there are little to no automated effluent monitoring solutions.

• As part of the Future Work, a technical discussion of the real-world deployment of these systems was also shown. This discussion identifies some of the practical obstacles associated with the development of a deployable monitoring system.

6.3 Future Work

There are a number of key areas of development for future work in this project.
6.3.1 Sample Handling

The technologies outlined in this thesis all require further development from a sample handling point of view. While this would not require significantly new technological development, it should not be considered trivial. The introduction of wash-rinse-dry cycles to vessels that may be left in-situ for weeks at a time requires careful design. A number of questions must be answered: Would the use of warm water introduce a condensation problem? How effective would this cleaning be? How often would a manual cleaning be required? Can a calibration image be used to remove effects of a dirty vessel from the image processing calculation?

How can all elements of the system be integrated into one unit? Can effluent quality be determined solely from the SSV supernatant liquid (this is dependant on the type of tertiary treatment), thereby reducing the number of interfaces with the WWTP? The camera sensor itself is separate from the sample medium, so it is protected from direct fouling by not being in direct contact with the sample. In a blacked-out environment the light source stability issue experienced during these tests should not be a problem.

6.3.2 Value Engineering – Cost Analysis

What is the potential cost associated with implementing this system? Numerous systems have been proposed and tested for wastewater monitoring; many of these have been developed on x86 architecture and also tested on lower end ARM proces-
sensors like the Raspberry Pi 3B. Conversion of the systems to non-proprietary operating environments helps reduce the cost of the potential system. The hardware requirements for the image processing system are minimal. Low end processing power – a consumer grade webcam is sufficient in terms of imaging. Lighting is provided by cheap generic white LEDs and low intensity IR LEDs. The results of these systems have proven their usefulness as measurement techniques and while they may not be perfect they are a great improvement over no measurement. While further development is required for full deployment of these proposed systems, they represent an important step forward in the search for robust monitoring solutions for wastewater treatment.

The remote monitoring system was developed using low-cost hardware and leveraged existing cloud storage services to provide a quickly deployable system capable of providing real-time updates of WWTP effluent turbidity. Further work is required in this area to scale the system for deployment to multiple sites that would be centrally managed. Data security is also important; therefore, integration with existing data management systems would be necessary.

6.3.3 System Integration with WWTP

The systems proposed in this thesis can generate a significant amount of data, based on the frequency of measurements. To facilitate distributed operation, a Service Oriented Architecture (SOA) would be suitable as the basis for the overall system.
architecture, whereby the core functionality of the system is exposed via cloud-based services. An SOA provides a platform independent, technology agnostic, and modular approach to the system’s construction. Additionally the system would be easier to extend and maintain over time, as well as supporting interoperability with other systems.

While these ICT-oriented requirements may appear to be rudimentary, the characterisation of the environment in which they are being enforced must be considered. For example, a decentralised and unmanned WWTP with low-bandwidth connection to the internet, will inevitably find it more difficult to employ an image-based real-time monitoring system to promote awareness of water quality and report on the facility’s operating efficiency, when compared to a centralised facility with the appropriate network infrastructure.

The internet connectivity characteristics of decentralised WWTPs (i.e. bandwidth constraints and deployment environment), as well as the specific data conventions and structure of the SVI data, means that there are no known existing solutions that could facilitate data collection [181].

To facilitate open data exchange between the interacting components in the system, open and standard protocols would be used throughout the system. In terms of communication protocols, TCP/HTTP would be used for streaming data between WWTPs and the cloud, and to encode these data streams JSON and SOAP/XML could be utilised. These components work together to deliver an automated data col-
lection mechanism for numerical measurements and binary files (e.g. image or CSV) from decentralised WWTPs. In simple terms, the system provides subject matter experts with timely access to data for monitoring and analysis, without incurring the logistical and technical costs associated with manual data collection.

A client-side PC, could be used to control the imaging hardware and transfer the data to a centralised cloud storage system. There is also a need for a local store of information. This local database operates as a redundant and alternative storage repository for data. Its primary role in the system is to facilitate storage of data from the client-side PC when there is no external connection available for transmitting data to the cloud. A cloud-based relational database would be used to store meta-data about all decentralised WWTPs, as well as persisting data that is transmitted from the client-side applications deployed on these sites. This would be a multi-tenancy database, enabling a single database to be used to store data across multiple sites.

### 6.3.4 Turbidity Analysis

Further development is needed for the ancillary systems to deal with loading, flushing and automatic cleaning of the vessel to ensure that calibration can be maintained over a long deployment. One potential implementation would be a flow cell with a purge and rinse cycle to avoid vessel fouling over time.

There is a minimum level of turbidity this system can measure; fluids with low
turbidity values (< 15 FAU) contain insufficient particles to scatter a measurable quantity of light energy towards the image sensor. A different light source with higher intensity could resolve this issue, or an imaging sensor with longer exposures times than used in this study. This addition would make this system more applicable to monitoring drinking water treatment plants, where much lower turbidity values are found than in wastewater.

There are two key areas for improvement of the current algorithm: (i) There is a potential for erroneous measurements due to spikes in the intensity profile as shown in Figure 5.2. (ii) The results showed a higher standard deviation of the CET measurements for turbidity values < 30 FAU.

A classification system may be required to define what images/CET values are outliers and remove them from the calculation. A CET could be classed as an outlier by examining the ROI of the image or the light intensity profile for discontinuities.

To automate this outlier removal, a classifier could be trained to disregard measurements where an incorrect value of exponential constant was chosen based on an aberration in the profile, potentially due to a bubble or large flocculate in the ROI, or an excessively noisy intensity profile. This process could be included in the algorithm in a number of ways. Image processing techniques could be utilised to identify large flocs or bubbles in the ROI and remove the results from the CET calculation. Potential solutions would include a particle analysis that bins the image based on a intensity threshold and then disregards an image that contains multiple large flocs.
or bubbles. Alternatively, multiple intensity profiles could be examined for discontinuities and their level of agreement computed, or a smoothing filter applied to the intensity profile to remove discontinuities. Increasing the number of images taken would yield more points of analysis. The mode of the CET measurements could be selected and the rest of the results removed from the calculation. The primary advantage of the proposed method is the potential for autonomous operation, though further work is required to implement such a system. The core technology proposed in Chapters 4 and 5 would form the basis for such a system.

6.3.5 SSV Analysis

There are a few areas for potential future work to make the proposed SSV imaging system more robust. The inclusion of blackout material around the test rig would improve the comparability between tests. While all artificial environmental light sources were turned off, the incident daylight was not completely removed from the system. Condensation on the vessel surface was also found to have an obscuring effect on the automated SSV measurements; it may be desirable to operate the camera in a dehumidified enclosure to guard against the impact of condensation as well as to ensure that moisture does not have any long term adverse effects on the camera. Minor discrepancies were present in samples that settle to less than 200 mL after 30 minutes. In these cases, the stirrer created a hollow in its wake that led to a non-uniform settled sludge surface. Since the camera was positioned at the midpoint
of the vessel, it had a view down on this surface. Therefore, the volume was slightly under- or over-estimated – depending on the intensity of the image in this area.

A feasibility study with lower-end hardware could be undertaken by examining the effects of compression on the images. The DSLR camera used for this study is above the required specification. The limits of scaling and compression could be examined to find the minimum required specification for the imaging system.

6.4 Conclusion

There are several well-known benefits that can be attributed to decentralised WWTPs, such as reduced costs relating to infrastructure, improved water quality and availability, and reduced conventional pollutants and emerging contaminants. However, given the geographical distribution and isolated nature of many of these plants, they can prove difficult to monitor in real-time. In this study, two performance monitoring systems were proposed for WWTPs; (i) a remote monitoring system for inspecting treated effluent was proposed and turbidity was estimated using a subjective test and image processing. (ii) A separate system for measuring the SSV was designed and tested using a variety of mixed liquor samples from local WWTPs. Both systems aim to improve the availability of information, such as turbidity and SSV, that could be used to improve process control, particularly in remote, small and medium scale treatment plants, where there is less regular monitoring due to lack of full-time staff. Our legislators and administrators are the people who must
ultimately decide on the timing and the nature of improved management of WWTPs.
Bibliography


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