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Abdurrahman Abdulkareem Aliyu
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Modeling Biological Nutrients Removal for Waste Water Treatment Plant Management: Case Study Ain El Houtz WWTP

Defended on 22/04/2024 Before the Following Committee:

Chair	Prof. Bensaoula Fouzia	University of Tlemcen, Algeria
Supervisor	Prof. Nadia Badr ElSayed	Alexandria University, Egypt
Co-Supervisor	Prof. Cherifa Abdelbaki	University of Tlemcen, Algeria
External Examiner	Dr. Oumou DIANCOUMBA	Ecole Normale Superieure De Bamako
Internal Examiner	Dr. Lotfi Benadda	University of Tlemcen, Algeria

PAN AFRICAN UNIVERSITY FOR WATER AND ENERGY SCIENCES (INCLUDING CLIMATE CHANGE)

Modeling Biological Nutrients Removal for Waste Water Treatment Plant Management: Case Study Ain El Houtz WWTP

A master's thesis submitted to the Pan African University Institute for Water and Energy Sciences (Including Climate Change) in partial fulfilment of the requirements for the award of Masters of Science degree in water sciences (Engineering Track).

Presented By

Abdurrahman Abdulkareem Aliyu

(B.Eng Water Resources and Environmental Engineering)

Supervisor: Prof. Nadia Badr ElSayed

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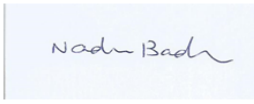
THESIS APPROVAL PAGE

Modeling Biological Nutrients Removal for Waste Water Treatment Plant Management: Case Study Ain El Houtz WWTP

Submitted by:

		
Abdurrahman Abdulkareem Aliyu		**22 – 03 – 2024**
Name of Student	**Signature**	**date of submission**
		
Prof. Nadia Badr ElSayed		**22 – 03 – 2024**
Name of Supervisor	**Signature**	**date of submission**

DEDICATION

To my lovely parents, Alh. Abdulkareem and Hajia Maryam
and sister, Habeeba
For your unwavering love, unending support, and continuous prayers

Proof that even with a healthy dose of procrastination, endless YouTube Shorts, and existential dread, dreams can come true (or at least this thesis can be finished)

STATEMENT OF THE AUTHOR

I, Abdurrahman Abdulkareem Aliyu, hereby declare that the thesis titled "Modeling Biological Nutrients Removal for Waste Water Treatment Plant Management: Case Study Ain El Houtz WWTP" is my original work. I have not submitted this work to any institution of higher education for the award of a degree, diploma, or certificate. I have followed all regulations of the Pan African University (PAU) Scholarship, and all the words and ideas borrowed from other works presented in this thesis have been appropriately cited and referenced according to academic rules and regulations. I have made every effort within my abilities to avoid plagiarism.

Student:

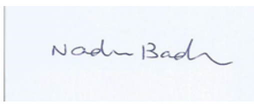
Name: Abdurrahman Abdulkareem Aliyu Signature: 

Date: 22/03/2024

Academic Unit: Water (Engineering Track)

PAU Institute: PAUWES

SUPERVISOR:

Name: Prof. Nadia Badr ElSayed Signature: 

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Institution: University of Alexandria, Egypt

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ACRONYMS AND ABBREVIATIONS

ASMs – Activated Sludge Models

ASM 1 - Activated Sludge Model No. 1

ASM 2 - Activated Sludge Model No. 2

ASM 2d - Activated Sludge Model No. 2d

ASM 3 - Activated Sludge Model No. 3

ASPs – Activated Sludge Processes

BOD_5 - Biological Oxygen Demand

BOD_u - Biological Oxygen Demand ultimate

BNR – Biological Nutrient removal

CBOD – Carboneous biological oxygen demand

COD - Chemical Oxygen Demand

DO – Dissolved Oxygen

EBPR – Enhanced Biological Phosphorus Removal

FAO – Food and Agriculture Organization

IAWPRC - International Association on Water Pollution Research and Control

IWA – International Water Association

MAE – Mean absolute error

MENA – Middle East and North Africa

NH_4-N – Ammonium Ion

N-NO_3^- / NO_3^-N - Nitrate-Nitrogen

N-NO_2^- / NO_2^-N - Nitrite-Nitrogen

N_2 – Nitrogen gas

N – Nitrogen

ONA - Office National de l'Assainissement

PAO – Phosphate Accumulating Organisms

P – Phosphorus

PO_4^{3}- - Phosphate

R.E – Removal Efficiency

RMSE – Root mean square error

S.D - Standard Deviation

TKN – Total Kjeldahl nitrogen

TP – Total Phosphorus

TSS – Total suspended solid

VSS – volatile suspended solid

WWTP – Wastewater Treatment Plant

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ABSTRACT

In recent years, it has become more important for wastewater treatment plants (WWTPs) to monitor nutrient concentrations in their effluents in order to protect the environment and human health. This study focuses on the Ain El Houtz WWTP and aims to develop a comprehensive model that accurately represents its biological nutrient removal process. The goal is to simulate its performance and assess the model's predictability. Operational data was collected and analyzed over a period of three years, from 2020 to 2022, to characterize the water quality of influent and effluent. Physicochemical parameters such as Total Suspended Solids (TSS), Chemical Oxygen Demand (COD), Biochemical Oxygen Demand (BOD_5), Ammonium-Nitrogen (NH_4), Nitrite-Nitrogen (N-NO_2-), Nitrate-Nitrogen (N-NO_3-), and Phosphate (PO_4-^{3}) were evaluated. Using the GPS-X modeling platform, the study developed a process flow diagram that integrates the ASM2d model for biological nutrient removal. Through sensitivity analysis, the research identified the key parameters that have an impact on nutrient removal efficiency, which in turn guided the calibration process. The focus of the calibration adjustments primarily lies on parameters associated with denitrification, autotrophic growth, and oxygen saturation coefficients. Statistical measures such as Mean Absolute Error (MAE) and Root Mean Squared Error (RMSE) were utilized to evaluate the model's performance in both steady-state and dynamic validation scenarios. Results indicated differences between simulated and observed concentrations for ammonium, nitrite, nitrate, and phosphate, underscoring the complexity of accurately modeling nutrient removal processes.

بِسْمِ ٱللَّهِ ٱلرَّحْمَٰنِ ٱلرَّحِيمِ

﴿ أَوَلَمْ يَرَ الَّذِينَ كَفَرُوا أَنَّ السَّمَاوَاتِ وَالْأَرْضَ كَانَتَا رَتْقًا فَفَتَقْنَاهُمَا ۖ وَجَعَلْنَا مِنَ الْمَاءِ كُلَّ شَيْءٍ حَيٍّ ۖ أَفَلَا يُؤْمِنُونَ(30)﴾

سورة : الأنبياء - 'Al-Anbiyā - الجزء : (17) - الصفحة: (324)

Have not those who disbelieve known that the heavens and the earth were joined together as one united piece, then We parted them? And We have made from water every living thing. Will they not then believe?

CHAPTER 1: INTRODUCTION

1.1 Background

Over the last two decades, increasing the nutrient concentrations in discharging effluents have become a major concern in the design and operation of wastewater treatment plants that necessitating the development of different treatment methods, such as physical, chemical, and biological. This deal with nutrients control and removal from the discharged system (Al-Rekabi, 2015) as the presence of high concentrations of phosphorus and nitrogen in wastewater causes several problems, such as eutrophication, oxygen consumption, and toxicity.

Nitrogen plays an important role in synthesizing proteins, nucleic acids, and various cell components, making it a vital nutrient in the biosphere. It forms a significant component of fertilizers and foods. However, it can lead to environmental issues when improperly treated. Large amounts of reactive nitrogen in receiving waters can cause problems such as toxic algae blooms, groundwater contamination, and the release of atmospherically active gases contributing to global warming (Winkler & Straka, 2019). Likewise, Phosphorus, in significant quantities in water, can lead to pH changes and variations in water oxygen levels, that causes harm to aquatic life. High concentration of phosphorus and nitrogen in receiving water are responsible for eutrophication process (Ramasahayam et al., 2014)

The primary goal of wastewater treatment has always been to protect human health, but in recent years, there has been a growing emphasis on protecting the environment as well. This is due to the realization that nutrient pollution can have a significant negative impact on aquatic ecosystems (Kehrein et al., 2020) . Nitrogen is known to exist in different oxidation states in wastewater, which makes the removal process complex and challenging. Among other removal methods, such as adsorption and co-precipitation, biological approaches have been known to effectively remove nitrogen compounds in wastewater. The activated sludge process, is considered as the most common biological wastewater treatment method that was developed to enhance the effectiveness of nutrient removal. In a conventional nitrogen removal process, wastewater goes through the nitrification and denitrification processes. Nitrification is the biological oxidation of ammonia or ammonium to nitrite, followed by nitrite oxidation to nitrate. However, denitrification reduces nitrate, producing N_2 through a series of intermediate gaseous nitrogen oxide products (Rahimi et al., 2020). In case of Phosphorus, the major physical forms of it in water bodies are divided into insoluble and soluble forms, while, according to its chemical properties, it can be divided into organophosphorus and inorganic phosphorus. Most of these are colloidal and granular, while inorganic phosphorus is always in the form of soluble phosphate, including orthophosphate and polyphosphate (Li et al., 2021). Phosphorus removal from wastewater is usually done using physicochemical methods, biological treatment, and/or combinations of both (Bunce et al., 2018).

Mathematical models are important tools for wastewater treatment research and development and for designing and optimizing the different treatment processes (Seco et al., 2020). In the past two decades, dynamic mathematical modeling has become increasingly popular in the wastewater treatment field. This allows researchers to predict the performance of reactors and simulate complex processes such as biodegradation, crystallization, adsorption, and filtration (Duan et al., 2019). Currently, plant-wide wastewater treatment models are developed that integrate different model extensions to better represent the actual processes that occur in wastewater treatment plants. These models also incorporate new concepts and technologies related to the circular economy, such as nutrient recovery and reuse (Seco et al., 2020).

Activated sludge models have gained widespread usage for interpreting biological performance and implementing plant-wide process optimization. These models have been utilized in activated sludge treatment plant design and control for nutrient removal from domestic sewage (H. G. Insel et al., 2009). For engineering practice, these models have been incorporated into commercial simulation packages such as BioWin, GPS-X, WEST and DESAS (Ruano et al., 2012)

The GPS-X software, a comprehensive plant-wide model, developed by Hydromantis Environmental Software Solutions Inc.(*GPS-X Technical Reference - v8.0*, 2019), which is used in this study, is quite popular among the other modeling software because of its wide variety of pre-built treatment technologies, easy to use, and has readily available training materials (Mu'azu et al., 2020). The software employs the "Mantis" models, a modified version of Activated Sludge Models (ASMs) developed by the International Water Association (IWA). Besides the kinetic description of the treatment process carried out at the WWTP, these models allow the simulation of new scenarios for the study of critical parameters for the process and optimization and control of the WWTP. GPS-X's WWTP design development required gathering physical data on unit processes and historical data on the plant's performance. (Sonaje & Berlekar, 2015)

1.2 Problem Statement

Ensuring sufficient access to water resources is crucial for sustainable progress as global freshwater is limited. Climate change, population growth, and economic expansion have contributed to this problem. The MENA region, particularly Algeria, faces severe water scarcity due to its arid climate. Irregular rainfall patterns and climate change-induced droughts are considered as a big challenge for water availability and management. The agricultural sector in Algeria, a major water consumer, suffers due to decreased water availability. To address the water shortage, farmers in some regions has resulted to using treated wastewater as an alternative source of irrigation water. However, ensuring that wastewater quality is suitable for agricultural purposes and does not contain contaminants that could pose risks to crops or public health is important (Moussaoui et al., 2023).

The challenges of water scarcity, stringent environmental regulations, and the need for sustainable agricultural practices, have necessitated renewed effort on efficient and cost-effective nutrient removal from wastewater in the Ain El Houtz WWTP. Utilizing treated wastewater for irrigation offers a promising solution to water scarcity challenges, but only if the treated effluent meets strict quality standards to protect human health and safeguard agricultural productivity. Mathematical modeling has proven to be a cost-effective method to understand and optimize the treatment process to meet the required effluent standard.

Although models are not the final solution for biological nutrient removal, they offer a balance between complexity and simplicity, which are essential tools for various purposes such as research (testing results or optimizing experiments), process optimization, troubleshooting at full-scale treatment plants, and design assistance (for optimizing details). For example, optimizing modern nutrient removal plants is very complicated due to the presence of many interacting processes. Therefore, a model would be a valuable tool for optimizing the operation, evaluating and implementing new procedures (Topkaya, 2008).

1.3 Research Questions

The main aim of this research is to address several sub-questions that arise in developing a model of the biological nutrient removal process at the Ain El Houtz WWTP, as follows:

a) What are the key operational parameters that affect the efficiency of the nutrient removal process at the Ain El Houtz WWTP?

b) How the ASM2d model will be applied for modeling biological nutrient removal for the WWTP system?

c) How can the results obtained from the modeling support decision-makers in making environmentally sustainable and cost-effective decisions to increase the efficiency of operational parameters of the Ain El Houtz WWTP?

1.4 Objectives of The Study

1.4.1 General Objective

This research uses GPS-X Software to develop a model for the biological nutrient removal process at the Ain El Houtz WWTP.

1.4.2 Specific Objectives

1. To develop a model that is representative of the Ain El Houtz WWTP using operational data (such as flow rate, pH, temperature, ammonia concentrations, phosphorous concentrations and other relevant parameters).

2. To identify key parameters influencing the biological nutrient removal process in the Ain El Houtz WWTP using sensitivity analysis.

3.To compare the results of the simulation model and experimental data and evaluate the predictability of the model.

1.5 Significance of the Study

This study highlights the important aspects of ensuring that the treated effluent's quality meets stringent standards set by the Food and Agriculture Organization (FAO) and the Algerian government for wastewater to be reused for irrigation. By assessing operational parameters and utilizing the GPS-X software for the modeling of biological nutrient removal, the research aims to support decision-makers in making environmentally sustainable and cost-effective choices.

CHAPTER II: LITERATURE REVIEW

2.1 Wastewater Treatment

Wastewater is composed of 99.9% water, with the remaining 0.1% consisting of suspended and dissolved organic and inorganic solids, such as macronutrients and essential micro-nutrients. The solid substances found in wastewater are a combination of detergents, food scraps, fats and oils, grease, heavy metals, various biomolecules along with their decomposition products, sand, grit, excrement, paper products, and several natural and synthetic organic chemicals that come from different process industries (Machineni, 2019). It also contains various harmful microorganisms, which can produce unpleasant odors. Additionally, wastewater contains large amounts of carbohydrates, lipids, phosphorus, and nitrogen, which can cause excessive growth of aquatic plants and algae, leading to poor water quality and a phenomenon known as eutrophication (Machineni, 2019).

Wastewater treatment refers to a series of processes designed to improve water quality by reducing the concentration of pollutants. The objective of this process is to meet the necessary health and environmental standards required for the treated water. Different treatment techniques are employed to eliminate suspended solids, which can contaminate rivers and obstruct the flow of water in pipes and channels after deposition (Jasim, 2020).

Wastewater treatment systems are designed to imitate natural processes for water purification, such as biological, physical, and chemical processes. The differences between these facilities are determined by the type of technology used, its intensity, and the possible combinations of technologies. All processes involved in wastewater treatment can be defined in terms of physical, chemical, biochemistry (including microbiology), and the speed of the process (Salgot & Folch, 2018). It often involves a biological process where microorganisms are used to break down biodegradable organics into more stable substances and biomass. This process can be categorized based on the method used to introduce microorganisms, which can either be attached or suspended growth. In the attached growth method, microorganisms are attached to a surface to create a biofilm as seen in trickling filters and rotating biological contactors (RBC). In the suspended growth method, microorganisms are present in a fluid, as seen in the conventional activated sludge (CAS) process and membrane bioreactors (MBR) (Waqas et al., 2020).

The process of biological treatment can be divided into two categories: aerobic and anaerobic processes. In aerobic treatment, microorganisms (aerobes) use free or dissolved oxygen to convert organic waste into biomass and CO_2. On the other hand, anaerobic treatment involves breaking down complex organic waste into methane, CO_2, and H_2O through three steps (hydrolysis, acidogenesis including acetogenesis, and methanogenesis) without oxygen (Chan et al., 2009). Aerobic treatment is commonly used to treat organic wastewater as it achieves high treatment efficiency. Meanwhile, anaerobic treatment is focused on resource recovery and utilization while still achieving the objective of pollution control. Significant progress has been made in anaerobic biotechnology for waste treatment (Chan et al., 2009).

2.2 Activated Sludge Process

In 1914, English scientists Ardern and Lockett made significant progress in the field of wastewater treatment with the introduction of the "activated sludge" system. This system involved reusing the "suspension" formed during the aeration process, which had previously yielded unsatisfactory results. According to Banadda et al., (2011), the activated sludge system is a biotechnological process that uses a mixture of bacterial cultures to convert contaminants in wastewater. It has proven to be a durable technology that is flexible, robust, and capable of meeting even the most stringent effluent criteria. The activated sludge process is the most widely used method for the secondary treatment of municipal wastewater. It is considered the fundamental process of many wastewater treatment plants (WWTPs). These plants are available in various sizes, from small package plants designed for single houses to large plants serving entire metropolitan areas (Makinia, 2010).

The activated sludge process is a method that encourages the growth of heterotrophic microbial biomass. The microorganism in the aeration tank helps to either assimilate or oxidize dissolved organic matter in influent wastewater, which is then converted into solids that can easily settle. These settled solids are removed by sedimentation, and some are recycled back into the system. The microorganisms that are not able to settle are drained out of the system, which helps maintain a sufficient concentration of microorganisms in the aeration tank. This concentration is essential for continuously breaking down the organic matter from the influent. The secondary settler plays a crucial role in providing enough time for the microorganisms to settle down (Najar & Naci Engin, 2019; Sheik et al., 2014)

Controlling the treatment process of wastewater is not as simple as it appears. This is because there are several factors that can affect it, such as changes in the combination of microorganisms in the treatment tanks and changes in the inflow of wastewater to the plant. The inflow of wastewater can vary in flow rate, chemical composition, pH, and temperature depending on the season and the time of day. Additionally, many wastewater treatment plants experience surge flows of rainwater following storms, which are usually managed through a bypass system. Some plants receive industrial wastewater, which can harm microorganisms and prevent the proper function of activated sludge (Ahansazan et al., 2014).

In the past hundred years, the activated sludge process has undergone various operational changes to enhance its efficiency and flexibility. Initially, the system was created to eliminate carbonaceous organic compounds and toxic ammonia that is harmful to fish. However, nowadays, many plants are also designed to eliminate other nitrogen and phosphorus-containing compounds through microbiological means (Seviour & Blackall, 1998).

2.2.1 Microorganisms in Activated Sludge Process

The aeration tank's activated sludge is a complex ecosystem of competing microorganisms in a wastewater treatment plant (Ahansazan et al., 2014). These microorganisms include bacteria, fungi, protozoa, and higher forms of animals such as rotifers, insect larvae, and worms (L. K. Wang et al., 2009); the most prevalent microorganisms are bacteria and protozoa (Ahansazan et al., 2014; Seviour & Blackall, 1998). Some of the bacterial species present in the activated sludge including pseudomonas, Zooglea, Bacillus, Athrobacter, Microthrix, Nocardia, Acinetobacter, Nitrosomonas, Nitrobacter and Achromobacter. Each of the microorganisms in such a mixed population has different growth patterns and rates. Such growth patterns depend on various factors such as food, nutrients, pH, temperature, and oxygen availability (Mayhew & Stephenson, 1997).

In a wastewater treatment plant, the bacteria present can be classified into two types: heterotrophy and autotrophy. The dominant group of organisms is the heterotrophic or carbonaceous bacteria, which predominantly rely on organic carbon molecules for their nutrition. Conversely, the autotrophs take in inorganic chemicals and use them to synthesize organic compounds. The most significant subgroup of autotrophs is the nitrifying bacteria, responsible for removing ammonia from the wastewater. Although there are relatively few species of autotrophs, they have low growth rates and tend to be outcompeted by the faster-growing heterotrophs (Ahansazan et al., 2014).

2.2.2 Oxidation in Activated Sludge Process

Activated sludge microorganisms require oxygen to break down organic matter and obtain energy for growth. If there is insufficient oxygen, it can slow down or even kill the aerobic microorganisms, causing the facultative microorganisms to work less efficiently. As the number of microorganisms in an aeration tank increases, so does the amount of oxygen needed to support them. High concentrations of BOD in the influent or a higher influent flow will increase the activity of the microorganisms, which in turn increases the demand for oxygen (Ahansazan et al., 2014). A sufficient amount of oxygen must always be maintained in the aeration tank to ensure waste stabilization. A minimum dissolved oxygen (D.O.) level of 1.0 mg/L is recommended in the aeration tank for most types of activated sludge processes. Maintaining a D.O. level of >1.0 mg/L helps to establish a favorable environment for the microorganisms, producing the appropriate level of activity. If the D.O. levels are too low, it can lead to insufficient oxygen for the microorganisms to treat the wastewater effectively. On the other hand, if the D.O. levels are too high, it can cause issues such as flock particles being floated to the surface of the secondary clarifiers, especially during cold weather. Therefore, it is crucial to maintain the proper dissolved oxygen levels in the aeration basin. This requires regular monitoring by the system operator using a D.O. meter (Ahansazan et al., 2014).

2.3 Biological Nutrient Removal in Activated Sludge Process

Biological processes for simultaneous nitrogen and phosphorus removal are widely used for treating domestic and industrial wastewater. One common process is the anaerobic, anoxic, and oxic (A/A/O) process, which falls under biological nutrient removal (BNR) (Fang et al., 2011). It has become an established technology in wastewater treatment practice due to the need to control eutrophication, which has been facilitated by an improved understanding of nitrification, denitrification, and excess biological phosphorus (P) removal (Hu et al., 2012)

Biological nutrient removal processes are used to eliminate nitrogen and phosphorus from wastewater by employing microorganisms under different environmental conditions. Although, biological uptake for the growth of biomass removes nitrogen and phosphorus, it is not significant enough to meet the required standards for domestic wastewater treatment. Therefore, other means are necessary. The primary biological processes that remove nitrogen are ammonification (which converts organic nitrogen to ammoniacal nitrogen), nitrification (which converts ammonia to nitrate), and denitrification (which converts nitrate to nitrogen gas (N_2)) which then escapes into the atmosphere (Sathasivan, 2009). For phosphorus removal, phosphate is released in the anaerobic tank and then taken up excessively by microorganisms (Phosphorus accumulating organisms (PAO)) in the subsequent aerobic tank. Thereby simultaneously removing nitrogen and phosphorus in the BNR process (Fang et al., 2011)

2.3.1 Nitrogen removal

Most of the nitrogen found in wastewater is in the form of ammonia (NH_4^+ and NH_3). The nitrogen is removed through the biological method, which is usually carried out in two processes. The first process is aerobic nitrification, where ammonium oxidizing bacteria (AOB) convert ammonia to nitrite (nitritation), and nitrite oxidizing bacteria (NOB) convert nitrite to nitrate (nitratation). The second process is denitrification, where denitrifies convert nitrate to dinitrogen gas using organic carbon as an electron donor (Winkler & Straka, 2019).

Nitrification is a two-step process that involves biological oxidation. In the first step, NH_4^+ is oxidized to NO_2^-, which is carried out by AOB. This step is catalyzed by ammonia monooxygenase (AMO) and hydroxylamine oxidoreductase (HAO), and hydroxylamine (NH_2OH) is formed as an intermediate product. In the second step, NO_2^- produced in the first step is quickly converted to NO_3^- in the presence of molecular oxygen, and this step is carried out by NOB. The conversion is catalyzed by nitrite-oxidoreductases (NXR) and nitrite-oxidizing systems, which are one-step oxidation enzymes found in Nitrobacter and genera of Nitrococcus, Nitrospina, and Nitrospira, respectively (Rahimi et al., 2020).

Denitrification process is facilitated by bacteria that convert nitrate (NO_3^-) to harmless nitrogen gas (N_2) in an anaerobic environment, undergoing a series of intermediate steps involving nitrite (NO_2^-), nitric oxide (NO), and nitrous oxide (N_2O) (McCarthy & Bronk, 2008). Denitrification offers a cost-effective approach to simultaneously treat contaminants, minimizing waste disposal expenses. This process requires strict anoxic conditions, carbon sources, and post-treatment for optimal performance. Additional organic carbon serves as the electron donor, fueling cellular growth and heterotrophic denitrification. Common carbon sources added to denitrification systems include glucose, alcohols like methanol and ethanol, and organic acids like succinate and acetate (Rahimi et al., 2020).

Another way to remove nitrogen from wastewater is the partial nitritation-anammox (PNA) method, which is an energy-saving advancement over conventional nitritation/denitrification for nitrogen removal in wastewater treatment. PNA utilizes anammox bacteria, which directly convert ammonium to dinitrogen gas using nitrite (produced by ammonia-oxidizing bacteria during partial nitritation) as the electron acceptor. Current implementations achieve roughly 50% ammonium oxidation by ammonia-oxidizing bacteria, with the remaining half anaerobically oxidized by anammox. Compared to conventional biological nitrogen removal, PNA offers several key advantages (Winkler & Straka, 2019); 1) Fully autotrophic nitrogen removal, hence no need for organic carbon source, 2). Reduced energy demand, with a decreased aeration of about 60%, 3) Minimized sludge production by roughly 75%, 4). Reduction in greenhouse gas emissions as PNA produces neither CO_2 nor N_2O, unlike conventional biological nitrogen removal (Winkler & Straka, 2019).

The nitritation-denitritation process, also known as the "Nitrite Shunt," is a method that prevents the oxidation of nitrite to nitrate by nitrite-oxidizing bacteria (NOB). It allows for the reduction of the formed nitrite to dinitrogen gas by heterotrophic denitrification, which can decrease the organic carbon demand for total nitrogen removal by 40%. Additionally, it can save up to 25% of the aeration costs by avoiding nitrite oxidation. Some studies have shown that a combination of controlled aeration phase length and dissolved oxygen (DO) at 15°C resulted in nitritation-denitritation reactions under aerobic granular sludge in SBR, resulting in a total nitrogen removal efficiency of up to 95% (Sandip, 2017).

Many systems have been engineered and applied to advance nitrogen removal in wastewater treatment, based on the known reactions. There are two major types of microbial growth processes available in biological wastewater treatment: biofilm-based and granular sludge-based. However, recent developments have also moved towards a hybrid process, which uses both biofilm and sludge-based approaches to treat the wastewater, as shown in Fig 2.1 (Sandip, 2017).

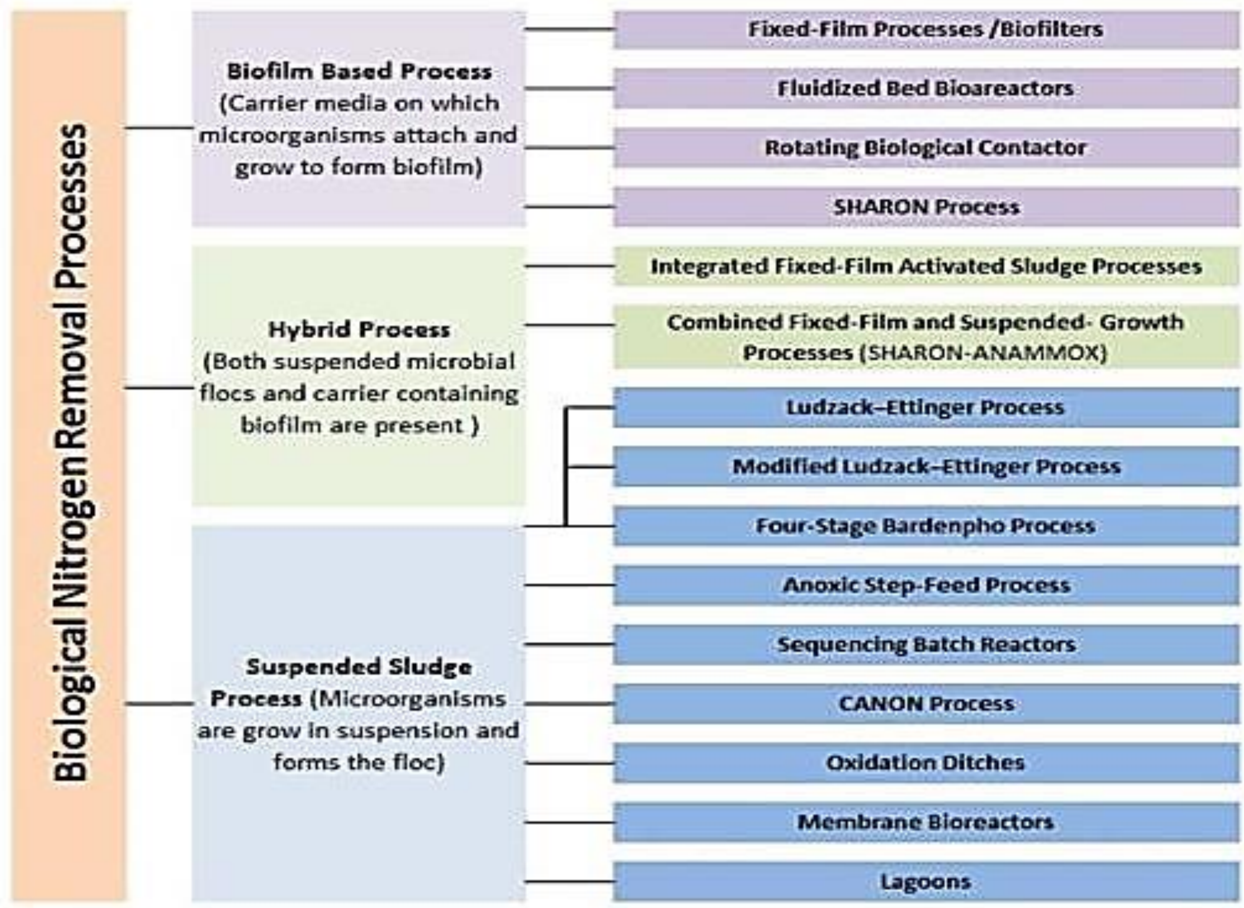


Figure 2. 1 The processes and technologies applied for biological nitrogen removal (Sandip, 2017)

2.3.2 Phosphorus Removal

Before discharging wastewater into water bodies, it is important to remove phosphate, even though, in many cases, it is not performed, leading to major contamination worldwide. The wastewater treatment industry uses several methods to remove phosphorus (De-Bashan & Bashan, 2004). The major treatment methods include adsorption, chemical precipitation, enhanced biological phosphorus removal, and constructed wetlands that are indirectly remove nutrients like phosphorus by utilizing it as nourishment for the plants, as they are not designed to specifically target nutrient removal. (De-Bashan & Bashan, 2004; Ramasahayam et al., 2014).

The adsorption method is a technique that uses an adsorption agent to attract phosphorus from a solution and separate it from the liquid phase for the purpose of removing it. Adsorbents have a large surface area and a strong ability to attract phosphorus in water, effectively removing it. Many low-cost minerals and industrial waste can be used as adsorbents, such as benthic soil, activated carbon, zeolite, and slag. The size of the adsorbent's aperture can greatly affect the adsorption volume and rate (Li et al., 2021).

Enhanced Biological Phosphorus Removal (EBPR) is a method used to remove phosphate from wastewater and convert it into sludge. This is achieved by using Polyphosphate-Accumulating Organisms (PAOs), which can take in more phosphorus than is necessary for their growth. The EBPR process can remove over 85% of phosphorus from wastewater (Ramasahayam et al., 2014). Under anaerobic conditions, PAOs remove organic carbon from the water and degrade intracellular glycogen and polyphosphate, releasing phosphate into the surrounding waters. Under aerobic conditions, the microorganisms take up phosphates and store them in the form of intracellular polyphosphate. Although some phosphorus is released back into the water under anaerobic conditions, there is still a net removal of phosphorus by these PAOs (Ramasahayam et al., 2014).

The chemical precipitation method aims to remove phosphorus from sewage by adding soluble chemical reagents into the water. These reagents react with the phosphate roots in the sewage, producing insoluble phosphates. This mixture is then separated from the water through solid separation. Lime and metal salts are common precipitants used for this process (Li et al., 2021). The principle behind lime precipitation is that calcium salts and phosphates in sewage react to form hydroxyphosphorus insoluble lime precipitation to remove phosphorus from the water. On the other hand, metal salts, such as iron salts and aluminum salts, work by reacting with phosphate roots in water to form large particles that are insoluble in water, ultimately removing phosphorus. Aluminum sulfate and polymerized aluminum chloride are commonly used aluminum salts, while iron salts include iron sulfate and iron chloride (Li et al., 2021).

Figure 2.2 outlines the different methods available for removing or converting phosphorus species. There are three options for removing phosphorous from the system: converting phosphorus to chemical species by adding a metal salt or lime (precipitation), removing it with membrane treatments, and incorporating the phosphorus into biomass (Sathasivan, 2009).

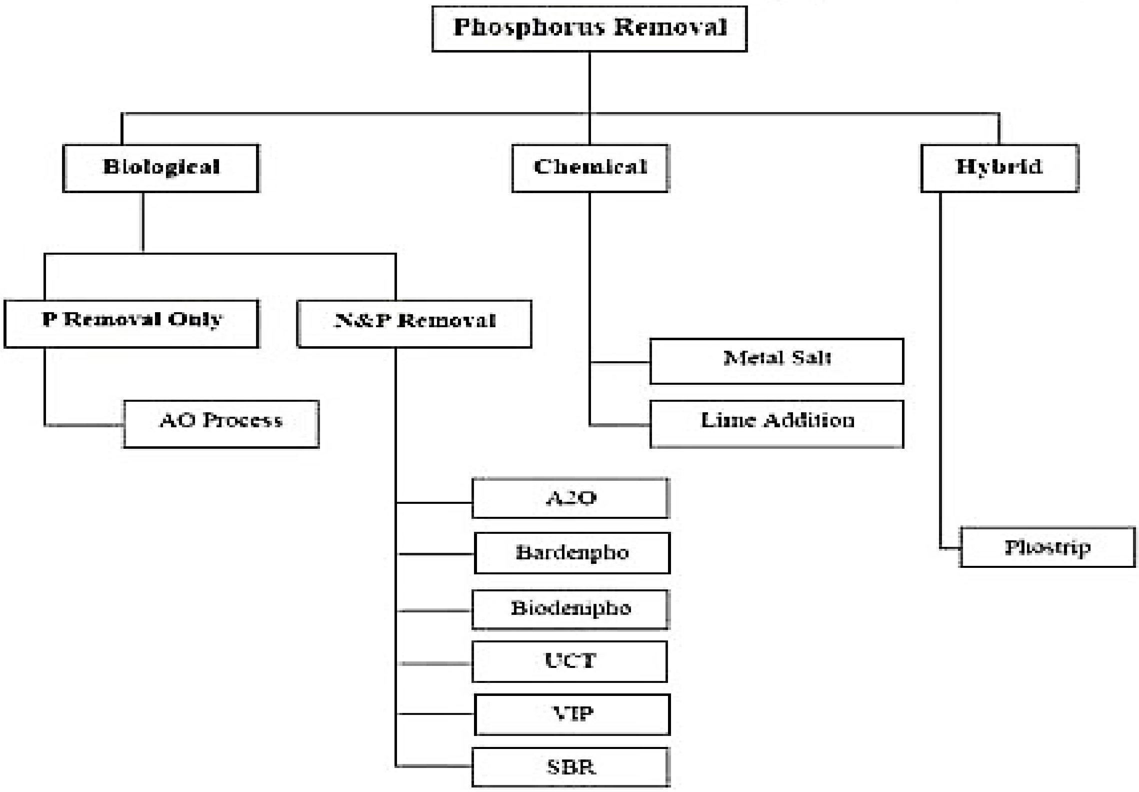


Figure 2. 2 Overview of Phosphorus removal process (Sathasivan, 2009)

2.4 Modeling and Simulation of Activated Sludge Process

Mathematical modeling of the activated sludge process is a powerful tool for designing, operating, forecasting future behavior, and controlling the processes (Nuhoglu et al., 2005). It has been an essential tool for simulating complex biochemical processes in wastewater treatment plants. The modeling requires data related to the wastewater's physical and operational parameters, sludge characteristics, process kinetics, and stoichiometry. Activated sludge models (ASM), such as those suggested by the International Water Association (IWA), are the most commonly used dynamic and steady-state mathematical expressions and models for designing, operating, and optimizing biological wastewater treatment plants (Mu'azu et al., 2020b). Since the early 1970s, significant efforts have been made to model the Activated Sludge Process (ASP). The IWA (previously IAWQ) task group has been developing Activated Sludge Models (ASMs) to achieve this objective. Today, the IWA models comprising ASM1, ASM2, ASM2d and ASM3 have proven to be excellent tools for modeling carbon oxidation, nitrification, denitrification, and biological phosphorus removal processes. (Nuhoglu et al., 2005).

Mathematical models are broadly classified into two types: mechanistic and empirical. Mechanistic models are developed based on the underlying knowledge of the biological, physical, or chemical components of a process. On the other hand, empirical models are derived from an analysis of experimental data. Mathematical models can also be classified as steady-state or dynamic. Steady-state models use constant values for input variables to predict constant values for output variables, whereas dynamic models predict the time-varying performance of a process (Gall, 1999).

The modeling of the activated sludge process can serve the following purposes (Sochacki, 2008):

- Planning tools for wastewater treatment plants are a crucial component of the overall water management system. However, it can be very complex and challenging to manage. An effective approach is to integrate the wastewater treatment plant model with models of the urban sewer system and the receiving waters. This integrated model can be successfully used to analyse various managerial options.
- Analysis of the operation of a wastewater treatment plant (WWTP) to identify any operational issues or plan for plant upgrades to improve performance.
- When designing new plants, it's common to face uncertainty about their future operational circumstances. To tackle this, traditional design rules are typically applied. However, with the help of a model, the performance of a new plant can be tested against the assumptions made during design, specifically related to the characteristics and variability of the plant influent. This allows for a more accurate assessment of the plant's potential performance under different operating conditions.
- For real-time plant control which can reduce operational costs and improve effluent quality.
- Research tools, which may be used to analyse operational features of treatment plants that are not well understood and not well described,
- For the training of plant operators for plant monitoring and operation.

2.4.1 Activated Sludge Model (ASM1)

In 1983, the IAWPRC established a task group with the objective of promoting practical models for wastewater treatment. The aim was to reach a consensus on a straightforward model that could accurately forecast the performance of single sludge systems carrying out carbon oxidation, nitrification, and denitrification (Henze et al., 1987). This resulted in a report produced on a modeling approach called Activated Sludge Model No. 1 (ASM1). It is a widely used model for describing wastewater treatment processes, particularly for activated sludge plants with biological nitrogen removal. Despite the various modifications and extensions made to it over the years, it remains the "state-of-the-art" model for describing wastewater treatment processes worldwide (Sochacki, 2008).

2.4.1.1 ASM1 components

In the ASM1 model, wastewater is defined by seven dissolved components (represented by the symbol S) and six particulate components (represented by the symbol X). These components are used to describe two types of biomass, seven organic matter fractions, and four nitrogenous fractions. Additionally, the model takes into account the dissolved oxygen concentration and alkalinity as two other important wastewater characteristics (Sochacki, 2008).

2.4.1.2 COD components in ASM1

The selection of Chemical Oxygen Demand (COD) by the committee was deemed the most appropriate parameter for characterizing carbon substrates due to its ability to establish a connection between electron equivalents present in the organic substrate, biomass, and utilized oxygen. Within the Activated Sludge Model (ASM 1), COD is categorized based on (1) solubility, (2) biodegradability, (3) biodegradation rate, and (4) viability (biomass) (Petersen et al., 2003):

- The total COD is segregated into soluble (S) and particulate (X) components.
- Further subdivision of COD includes non-biodegradable organic matter and biodegradable matter. Non-biodegradable matter, biologically inert, passes through an activated sludge system unchanged. Inert soluble organic matter (Su) exits the system at the same concentration as it enters. Inert suspended organic matter in wastewater influent (Xu) or produced via decay (Xp) becomes entangled in activated sludge and is removed from the system through sludge wastage.
- Biodegradable matter is categorized into soluble readily biodegradable (Ss) and slowly biodegradable (Xs) substrate. Notably, some slowly biodegradable matter may be soluble. Readily biodegradable substrate is assumed to comprise relatively simple molecules that heterotrophic organisms can directly assimilate for biomass growth.

Conversely, slowly biodegradable substrate consists of complex molecules requiring enzymatic breakdown before utilization.

- Heterotrophic biomass (XBH) and autotrophic biomass (XBA) are produced through growth on readily biodegradable substrate (Ss) or on ammonia nitrogen (SNH). Biomass undergoes loss through the decay process, converting to Xp and Xs

- The total balance of the COD component in ASM1 is illustrated in Fig. 2.3

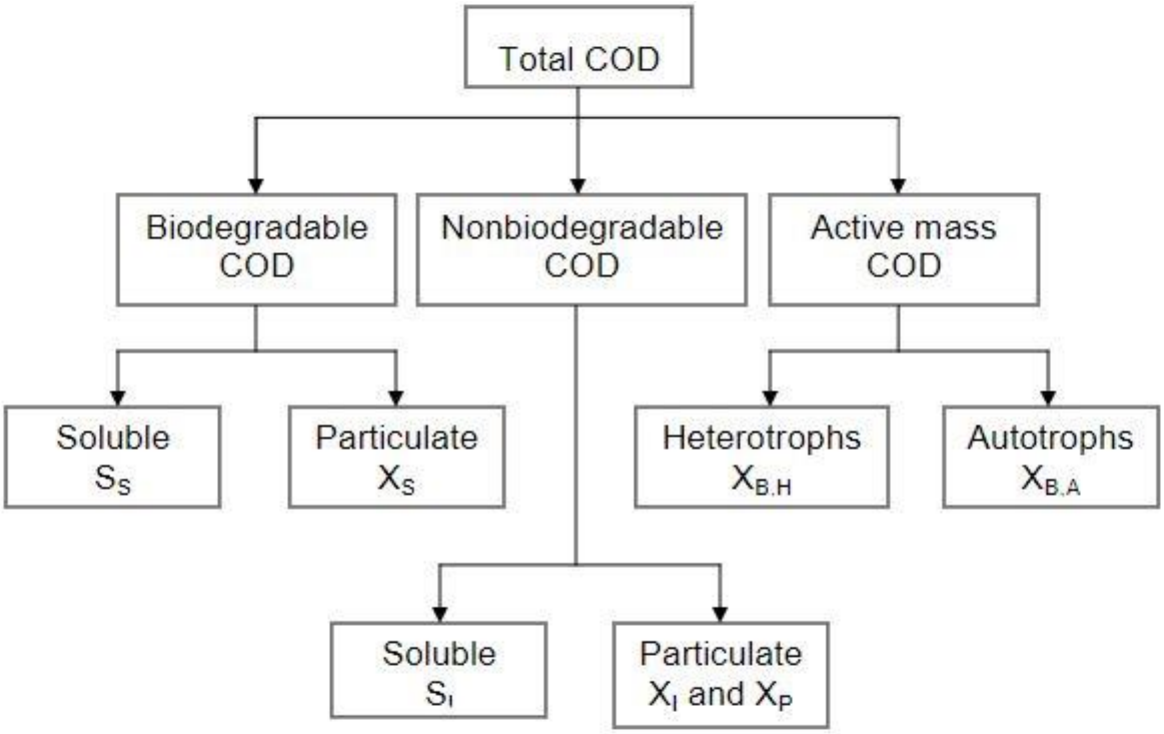


Figure 2. 3 Component of COD in ASM 1 (Sochacki, 2008).

2.4.1.3 Nitrogen components in ASM1

In a manner similar to the organic matter, total nitrogen can be categorized based on (1) solubility, (2) biodegradability, and (3) biodegradation rate (Petersen et al., 2003):

- Total nitrogen is segregated into soluble (S) and particulate (X) components.

- Nitrogen is further divided into non-biodegradable and biodegradable components. Non-biodegradable particulate organic nitrogen (XNI) is associated with non-biodegradable particulate COD (XI or Xp), while soluble non-biodegradable organic nitrogen (SNI) is considered negligible and thus excluded from the model.

- Biodegradable nitrogen is subdivided into ammonia nitrogen (Srm), nitrate + nitrite nitrogen (SNO), soluble organic nitrogen (SND), and particulate organic nitrogen (XND). The particulate organic nitrogen undergoes hydrolysis to soluble organic nitrogen concurrently with the hydrolysis of slowly biodegradable organic matter (Xs), present in the wastewater or produced through decay. Soluble organic nitrogen undergoes conversion to ammonia- nitrogen through ammonification. Ammonia nitrogen serves as the nitrogen source for biomass growth (parameter iXB denotes the nitrogen amount incorporated per COD unit). Ultimately, autotrophic conversion of ammonia yields nitrate- nitrogen (SNO), treated as a single-step process in ASM1.

The total balance of the nitrogen component in ASM1 is showed in Fig. 2.4

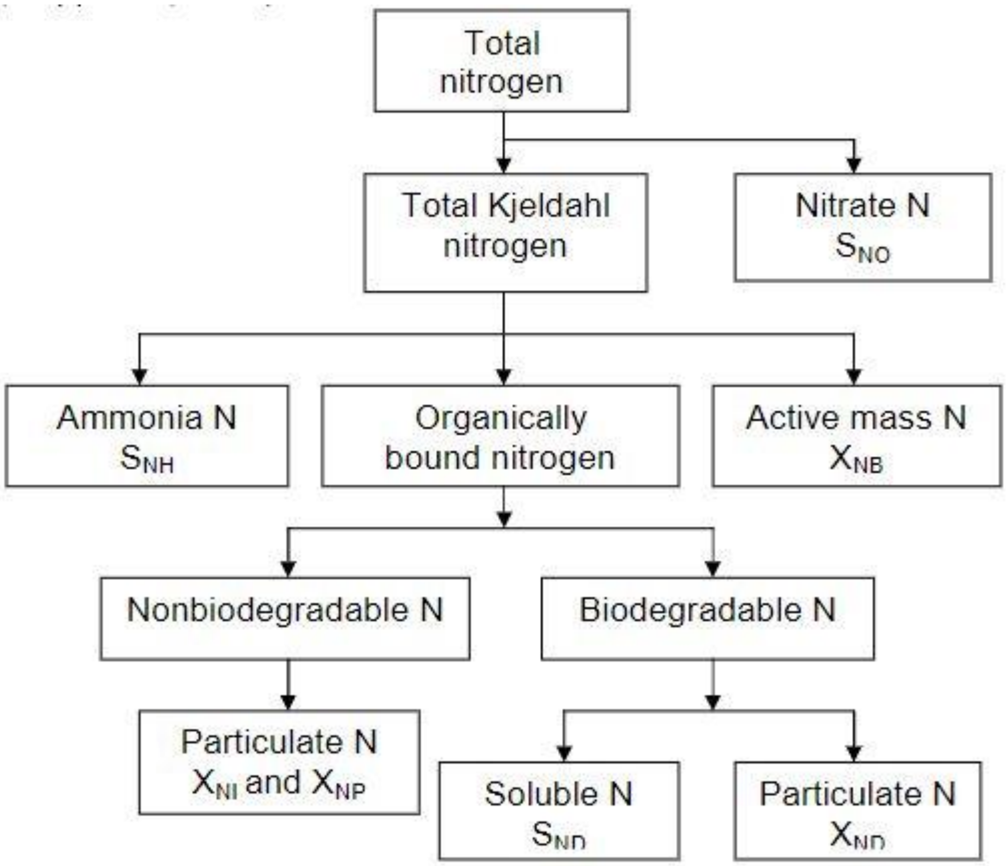


Figure 2. 4 Nitrogen component of ASM1 (Sochacki, 2008).

2.4.2 Processes in ASM1

The ASM1 model consists of eight processes, including three associated with biomass growth, two depicting biomass decay, two detailing the hydrolysis of particulate organics within the biofloc, and the remaining process addressing the ammonification of organic nitrogen. A process flow of the ASM1 is presented Fig 2.5 (Sochacki, 2008).

The various processes below are outline by Jeppsson (1996):

- ***Aerobic growth of heterotrophs***: A portion of the readily biodegradable substrate is utilized for heterotrophic biomass growth, while the remaining is oxidized for energy, resulting in an associated oxygen demand. Growth follows Monod kinetics, with ammonia serving as the nitrogen source for synthesis. Concentrations of SS and SO may limit the growth process, which primarily contributes to biomass production and COD removal, accompanied by an alkalinity change.

- ***Anoxic growth of heterotrophs***: In the absence of oxygen, heterotrophic organisms can utilize nitrate as the terminal electron acceptor with SS as the substrate, leading to heterotrophic biomass production and denitrification (formation of nitrogen gas). The process, governed by Monod kinetics, incorporates a reduction factor ηg (< 1) due to either reduced maximum growth rate under anoxic conditions or limited functionality of biomass with nitrate as the electron acceptor. Ammonia serves as the nitrogen source for cell synthesis, affecting alkalinity.

- ***Aerobic growth of autotrophs***: Ammonia undergoes oxidation to nitrate via nitrification, resulting in autotrophic biomass production and an associated oxygen demand. Ammonia is also used for synthesis, influencing alkalinity. The effect on biomass formation is minimal due to the low yield of autotrophic nitrifiers, with growth modeled using Monod kinetics.

- ***Decay of heterotrophs***: Modeled according to the death-regeneration hypothesis, where organisms perish at a certain rate, with a portion contributing to the X_P fraction and the rest adding to the pool of slowly biodegradable substrate. Organic nitrogen associated with X_S becomes available as particulate organic nitrogen, with no COD loss or electron acceptor utilization assumed, maintaining a consistent rate across aerobic, anoxic, and anaerobic conditions.

- ***Decay of autotrophs***: Modeled similarly to decay of heterotrophs.

- ***Ammonification of soluble organic nitrogen***: Biodegradable soluble organic nitrogen is converted to free and saline ammonia via a first-order process facilitated by active heterotrophs, resulting in an alkalinity change due to hydrogen ion consumption.

- ***Hydrolysis of entrapped organics:*** Slowly biodegradable substrate within the sludge mass is extracellularly broken down, yielding readily biodegradable substrate for organismal growth. Modeled based on surface reaction kinetics, occurring solely under aerobic and anoxic conditions. Anoxic conditions reduce the hydrolysis rate by a factor η_h (< 1), saturating as the entrapped substrate-to-biomass ratio increases.

- ***Hydrolysis of entrapped organic nitrogen***: Biodegradable particulate organic nitrogen is broken down into soluble organic nitrogen at a rate determined by the hydrolysis reaction for entrapped organics described above

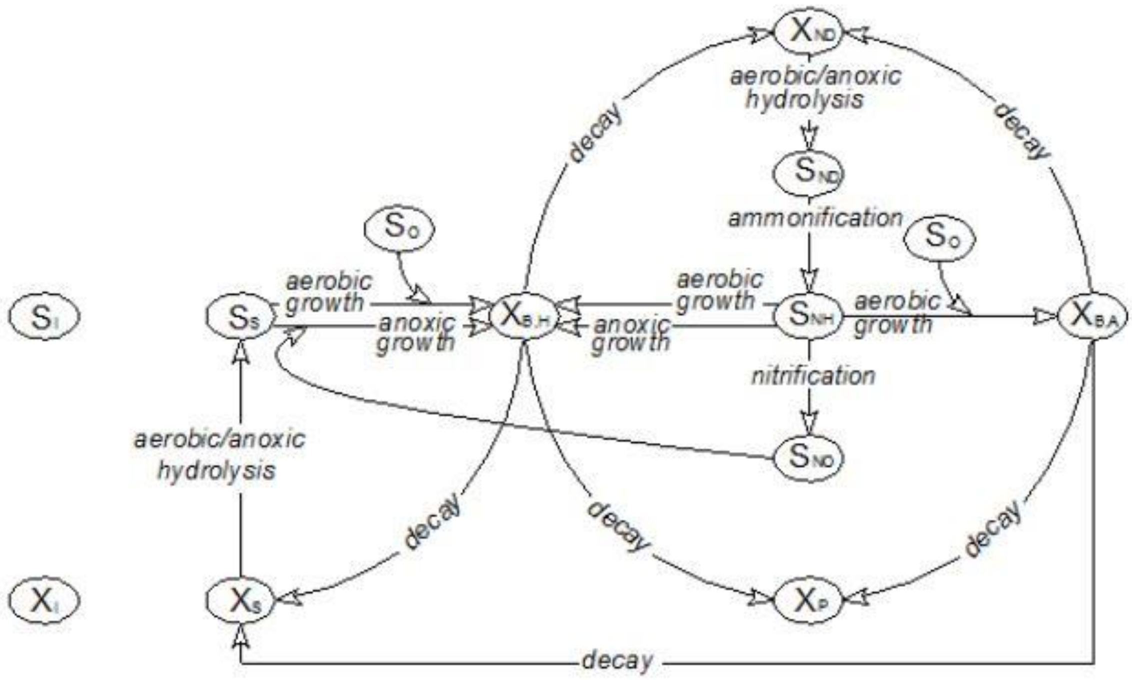


Figure 2. 5 The flow processes of the ASM1 (Sochacki, 2008).

2.4.3 Model Restriction of ASM1

Certain simplifications and assumptions are necessary when constructing a model of a wastewater treatment system. These limitations can pertain to both the physical system and the mathematical model. Below are summarized restrictions associated with ASM.1 (Jeppsson, 1996):

- The system is assumed to operate at a constant temperature. To accommodate temperature variations, an Arrhenius equation may be employed to adjust model parameters within a specified range.

- The hydrogen ion concentrations (pH) is considered constant and maintained near neutrality. The incorporation of alkalinity in the model enables the identification of potential issues with pH control.

- Changes in the nature of organic matter within specific fractions, such as the readily biodegradable substrate, are not considered. Coefficients in rate expressions are assumed to have constant values, making it challenging for the model to properly handle variations in wastewater characteristics.

- The model does not account for limitations of nitrogen, phosphorus, and other inorganic nutrients on the removal of organic substrate and cell growth. Care must be taken to ensure sufficient quantities of inorganic nutrients are present for balanced growth.

- Correction factors for denitrification are fixed and constant for a given wastewater.

- Coefficients for nitrification are assumed to be constant and to encompass any inhibitory effects that other waste constituents might have on them.

- Heterotrophic biomass is assumed to be homogeneous and does not undergo changes in species diversity over time. Effects of substrate concentration gradients, reactor configuration, etc., on sludge settleability are not considered.

- The entrapment of particulate organic matter in the biomass is assumed to be instantaneous.

- Hydrolysis of organic matter and organic nitrogen is coupled and occurs simultaneously at equal rates.

- The type of electron acceptor present is considered not to affect the loss of active biomass by decay.

2.5 Activated Sludge Model No.2 (ASM No. 2)

The publication of Activated Sludge Model 1 (ASM1) established a foundational framework for developing subsequent ASMs. However, ASM1 lacked the capability to simulate biological phosphorus removal, a treatment process already employed, although to a limited extent, in wastewater treatment plants (WWTPs). From the mid-1980s to the mid-1990s, the widespread adoption of biological phosphorus removal processes led to a deeper understanding of the underlying mechanisms. Consequently, ASM2 was introduced in 1995, incorporating nitrogen and biological phosphorus removal pathways (Hassan et al., 2020).

ASM2 included additional biological processes and components to characterize wastewater and activated sludge, particularly for biological phosphorus removal (Henze et al., 2000), which introduces Phosphorus Accumulating Organisms (PAOs) as a new group of microorganisms in the activated sludge, clarifying the ability of different microorganisms to accumulate phosphorus in the form of stored polyphosphates (Gujer et al., 1995).

However, there are limitations to the studies on the exact role of PAO, and therefore, some assumptions were made (Ghanesh, 2010):

1. PAOs are only able to use fermentation products like acetate for their growth.
2. PAO can only grow aerobically on stored PHA and cannot directly utilize fermentation products for growth.
3. PAO cannot denitrify
4. PAOs can store glycogen and carbohydrates as carbon reserves, but they are not yet included as a model parameter due to insufficient information.
5. Similar assumptions as in ASM1 with respect to pH and coefficient values.
6. Organic nitrogen and organic phosphate are coupled, just like in the ASM1 hydrolysis of organic matter.
7. Growth limitations at low inorganic nutrient concentrations were not considered, so it is necessary to assume that sufficient nutrients are provided.
8. PAO's reduced poly-phosphate uptake in the absence of cations like magnesium and potassium was not considered.
9. The inhibitory effect of nitrite and nitrogen monoxide was not considered
10. The behaviour of PAO was not studied beyond a temperature range of 10-25°C. Therefore, the model is only applicable to domestic wastewater.

2.6 Activated Sludge Model No.2d (ASM No. 2d)

The Activated Sludge Model No. 2d is an extension of ASM2 that includes two additional processes. These processes account for the fact that phosphorus-accumulating organisms (PAOs) can use cell internal organic storage products for denitrification. Unlike ASM2, which assumes that PAOs grow only under aerobic conditions, ASM2d includes denitrifying PAOs (Henze et al., 1999).

The symbols used to represent model components make a distinction between soluble "S" and particulate "X". In activated sludge systems, particulate components (X) are assumed to be associated with the activated sludge and can be concentrated by sedimentation or thickening in clarifiers. Soluble components (S), however, will only be transported with the water. It's important to note that all particulate model components (X) must be electrically neutral, with no ionic charges, while soluble components (S) may carry an ionic charge (Henze et al., 2000).

2.6.1 Definition of Soluble Components

The definition of the soluble components is provided below (Henze et al., 1999; Topkaya, 2008):

S_A: Fermentation products are considered to be acetate. Since fermentation is a biological process, these products must be modeled separately from other soluble organic materials. They are the end products of fermentation, and for all stoichiometric computations, it is assumed that S_A is equal to acetate. In reality, a range of other fermentation products dominated by acetate is possible.

S_{ALK}: Alkalinity of the wastewater. The alkalinity of wastewater is a measure used to estimate the continuity of electrical charges in biological reactions. It is introduced to determine the likelihood of low pH conditions that could hinder some biological processes. For all stoichiometric computations, it is assumed that S_{ALK} is bicarbonate, specifically H_{CO3}^{-}.

S_F: Fermentable, readily biodegradable organic substances. This fraction of the soluble COD can be directly available by heterotrophic organisms. It is assumed that S_F can serve as a substrate for fermentation, therefore it does not include fermentation products

S_I: Inert soluble organic material. The main feature of S_I is that these organic compounds cannot be broken down any further in the treatment facilities. This material is considered to be a part of the influent, and it is also assumed to be produced during the hydrolysis of particulate substrates X_S.

S_{N2}: Dinitrogen, N_2. It is assumed to be the only product of denitrification. It maybe subject to gas exchange, parallel with oxygen, S_{O2}-

S_{NH4}: Ammonium plus ammonia nitrogen. For the electrical balance of the electrical charges, S_{NH4} is assumed to be all NH4+.

S_{NO3}: Nitrate plus nitrite nitrogen. ($N_{O3}^{-} + N_{O2}^{-}$-N). S_{NO3} is assumed to include nitrate and nitrite nitrogen due to the absence of a dedicated model component for nitrite**.** For all stoichiometric computations (COD conservation), S_{NO3} is considered to be N_{O3}^{-}-N only.

S_{O2}: Dissolved oxygen. It may be subject to gas exchange.

S_{PO4}: Inorganic soluble phosphorus, primarily ortho-phosphates. For the balance of electrical charges, it is assumed that S_{PO4} consists of 50% $H_2PO_4^-$ and 50% HPO_4^{-2}, independent of pH.

S_S: Readily biodegradable substrate. This component was introduced in ASM1. In ASM2, it is replaced by the sum of SF + SA

2.6.2 Definition of Particulate Components

The definition of the particulate components is provided below (Henze et al., 1999; Topkaya, 2008):

X_{AUT}: Nitrifying organisms. Nitrifying organisms are responsible for nitrification; they are obligate aerobic, chemo-litho-autotrophic. It is assumed that nitrifiers oxidize ammonium directly to nitrate.

X_H: Heterotrophic organisms. These organisms are considered to be versatile heterotrophic organisms as they can grow both aerobically and anoxically (through denitrification) and can also be active anaerobically through fermentation. They are responsible for the hydrolysis of particulate substrates X_S and can utilize all degradable organic substrates under various environmental conditions.

X_I: Inert particulate organic material. This material does not degrade within the treatment systems. It may be a fraction of the influent or biomass decay.

X_{MeOH}: Metal-hydroxides. This component represents the ability of metal-hydroxides to bind with phosphorus, which can either be present in the wastewater or added to the system. For all calculations involving the chemical makeup, it is assumed that this component is made up of $Fe(OH)_3$. However, it is possible to replace this component with other reactants. Doing so would require adjusting the stoichiometric and kinetic information accordingly.

X_{MeP}: Metal-phosphate, MePO4. This component is the result of binding phosphorus to metal-hydroxides. In all stoichiometric computations, it is assumed that this component is made up of FePO4. If you want to use other precipitation products, you will need to adjust the stoichiometric and kinetic information accordingly.

X_{PAO}: Phosphate-accumulating organisms, PAO. These organisms are assumed to be representative of all types of poly-phosphate-accumulating organisms. The concentration of X_{PAO} only accounts for the 'true' biomass and does not include the cell internal storage products X_{PP} and X_{PHA}. In ASM2d, it is assumed that these organisms can grow in both anoxic and aerobic environments, whereas ASM2 only considers aerobic growth

X_{PHA}: A cell internal storage product of phosphorus-accumulating organisms PAO. It includes primarily poly-hydroxy-alkanoates (PHA) and is found exclusively in association with X_{PAO}. However, X_{PHA} is not considered a part of the X_{PAO} mass. It's worth noting that X_{PHA} cannot be directly compared to PHA concentrations that are measured analytically. X_{PHA} is only a functional component that is required for modeling purposes and cannot be chemically identified. That being said, it may be recovered during COD analysis, where it must satisfy COD conservation. In stoichiometric considerations, PHA is assumed to have the chemical composition of poly-β-hydroxy-butyrate $(C_4H_6O_2)_n$.

X_{PP}: Poly-phosphate. Poly-phosphate is an inorganic storage product found within cells of PAO (polyphosphate-accumulating organisms). It is only found in association with XPAO, but it is not included in the mass of XPAO. It is a component of the particulate phosphorus and can be observed analytically. For stoichiometric considerations, poly-phosphates are assumed to have the composition of $(K_{0.33}Mg_{0.33}PO_3)_n$.

X_S: Slowly biodegradable substrates. Slowly biodegradable substrates are high molecular weight, colloidal, and particulate organic materials that require cell external hydrolysis before they can be available for degradation. It is assumed that the hydrolysis products (SF) may be fermented.

X_{TSS}: Total suspended solids, TSS. Total suspended solids are included in the biokinetic models to calculate their concentration through stoichiometry. Since removal and precipitation of phosphorus introduce mineral fractions into the activated sludge, prediction of TSS is essential.

2.6.3 Typical Values for stoichiometric coefficients and kinetic parameters

According to Henze et al. (1999), It is possible that future experience and experimental pilot studies may result in different and better estimates for the model parameters. The user of ASM2 and ASM2d is responsible for determining the concentrations of relevant components in the wastewater, as well as the stoichiometric and kinetic parameters that apply to the specific case being dealt with. The Task Group provided a list of typical concentrations of model components in a primary effluent and a set of model parameters as a reference for testing computer code and as a first estimate for designing possible experiments to determine these parameters more accurately. However, this does not mean that ASM2 or ASM2d is reliable with these parameters in every case, nor that these parameters represent the state of the art. They were merely presented as a reference (Henze et al., 1999).

Tables 2.1 and 2.2 display a list of typical stoichiometric coefficients, and the definitions, and typical values of all kinetic parameters. The stoichiometric coefficients were based on previous experience with ASM1 or derived from verification trials of ASM2 concerning full-scale experience. Some kinetic parameters were estimated based on experience with ASM1, while those related to biological phosphorus removal were estimated based on laboratory experience and full-scale verification trials of ASM2 (Henze et al., 1999).

Table 2. 1 Definitions, and typical values of all kinetic parameters for ASM2d (Henze et al., 2000)

	Temperature	20ºC	10ºC	Units
	Hydrolysis of particulate substrates: X_S			
K_h	Hydrolysis rate constant	3.00	2.00	d-1
η_{NO3}	Anoxic hydrolysis reduction factor	0.60	0.60	-
η_{fe}	Anaerobic hydrolysis reduction factor	0.40	0.40	-
K_{O2}	Saturation/inhibition coefficient for oxygen	0.20	0.20	g O_2 m^{-3}
K_{NO3}	Saturation/inhibition coefficient for nitrate	0.50	0.50	g N m^{-3}
K_X	Saturation coefficient for particulate COD	0.10	0.10	g X_S g^{-1} X_H
	Heterotrophic organisms: X_H			

μ_H	Maximum growth rate on substrate	6.00	3.00	g X_S g^{-1} X_H d^{-1}
q_{fe}	Maximum rate for fermentation	3.00	1.50	g S_F g^{-1} X_H d^{-1}
η_{NO3}	Reduction factor for denitrification	0.80	0.80	-
b_H	Rate constant for lysis and decay	0.40	0.20	d^{-1}
K_{O2}	Saturation/inhibition coefficient for oxygen	0.20	0.20	g O_2 m^{-3}
K_F	Saturation coefficient for growth on SF	4.00	4.00	g COD m^{-3}
K_{fe}	Saturation coefficient for fermentation of SF	4.00	4.00	g COD m^{-3}
K_A	Saturation coefficient for growth on acetate SA	4.00	4.00	g COD m^{-3}
K_{NO3}	Saturation/ inhibition coefficient for nitrate	0.50	0.50	g N m^{-3}
K_{NH4}	Saturation coefficient for ammonium (nutrient)	0.05	0.05	g N m^{-3}
K_P	Saturation coefficient for phosphate (nutrient)	0.01	0.01	g P m^{-3}
K_{ALK}	Saturation coefficient for alkalinity (HCO3-)	0.10	0.10	mole HCO_3^- m^{-3}
	Phosphorus-accumulating organisms: X_{PAO}			
q_{PHA}	Rate constant for storage of X_{PHA} (base X_{PP})	3.00	2.00	g X_{PHA} g^{-1} X_{PAO} d^{-1}
q_{PP}	Rate constant for storage of X_{PP}	1.50	1.00	g X_{PHA} g^{-1} X_{PAO} d^{-1}
μ_{PAO}	Maximum growth rate of PAO	1.00	0.67	d^{-1}
η_{NO3}	Reduction factor for anoxic activity	0.60	0.60	-
b_{PAO}	Rate for lysis of X_{PAO}	0.20	0.10	d^{-1}
b_{PP}	Rate for lysis of X_{PP}	0.20	0.10	d^{-1}
b_{PHA}	Rate for lysis of X_{PHA}	0.20	0.10	d^{-1}
K_{O2}	Saturation/inhibition coefficient for oxygen	0.20	0.20	g O_2 m^{-3}
K_{NO3}	Saturation coefficient for nitrate, S_{NO3}	0.50	0.50	g N m^{-3}
K_A	Saturation coefficient for acetate SA	4.00	4.00	g COD m^{-3}
K_{NH4}	Saturation coefficient for ammonium (nutrient)	0.05	0.05	g N m^{-3}
K_{PS}	Saturation coefficient for phosphorus in storage of PP	0.20	0.20	g P m^{-3}
K_P	Saturation coefficient for phosphate (nutrient)	0.01	0.01	g P m^{-3}
K_{ALK}	Saturation coefficient for alkalinity (HCO_3^-)	0.10	0.10	mole HCO_3^- m^{-3}
K_{PP}	Saturation coefficient for poly-phosphate	0.01	0.01	g X_{PP} g^{-1} X_{PAO}
K_{MAX}	Maximum ratio of X_{PP}/X_{PAO}	0.34	0.34	g X_{PP} g^{-1} X_{PAO}
K_{IPP}	Inhibition coefficient for PP storage	0.02	0.02	g X_{PP} g^{-1} X_{PAO}
K_{PHA}	Saturation coefficient for PHA	0.01	0.01	g X_{PHA} g^{-1} X_{PAO}

	Nitrifying organisms (autotrophic organisms): X_{AUT}			
μ_{AUT}	Maximum growth rate of X_{AUT}	1.00	0.35	d^{-1}
b_{AUT}	Decay rate of X_{AUT}	0.15	0.05	d^{-1}
K_{O2}	Saturation coefficient for oxygen	0.50	0.50	g O_2 m^{-3}
K_{NH4}	Saturation coefficient for ammonium (substrate)	1.00	1.00	g N m^{-3}
K_{ALK}	Saturation coefficient for alkalinity	0.50	0.50	mole HCO_3^- m^{-3}
K_P	Saturation coefficient for phosphorus	0.01	0.01	g P m^{-3}
	Precipitation			
k_{PRE}	Rate constant for P precipitation	1.00	1.00	m^{-3} g^{-1} $Fe(OH)_3 d^{-1}$
k_{RED}	Rate constant for redissolution	0.60	0.60	d^{-1}
K_{ALK}	Saturation coefficient for alkalinity	0.50	0.50	mole HCO_3^- m^{-3}

Table 2. 2 Typical stoichiometric coefficients for ASM2d (Henze et al., 2000)

			Units
	Hydrolysis:		
f_{SI}	Production of SI in hydrolysis	0	g COD g^{-1} COD
	Heterotrophic biomass: X_H		
Y_H	Yield coefficient	0.625	g COD g^{-1} COD
f_{XI}	Fraction of inert COD generated in biomass lysis	0.10	g COD g^{-1} COD
	Phosphorus-accumulating organisms: X_{PAO}		
Y_{PAO}	Yield coefficient (biomass/PHA)	0.625	g COD g^{-1} COD
Y_{PO4}	PP requirement (PO4 release) per PHA stored	0.40	g P g^{-1} COD
Y_{PHA}	PHA requirement for PP storage	0.20	g COD g^{-1} P
f_{XI}	Fraction of inert COD generated in biomass lysis	0.10	g COD g^{-1} COD
	Nitrifying organisms: X_{AUT}		
Y_A	Yield of autotrophic biomass per NO_3^--N	0.24	g COD g^{-1} N
f_{XI}	Fraction of inert COD generated in biomass lysis	0.10	g COD g^{-1} COD

2.6.4 Limitations of ASM2d

The important limitations for ASM2d as documented by Henze et al. (2000) includes.:

- This model is applicable only to municipal wastewater.
- It is not possible to model processes that have an overflow of SA into the aeration tank.
- The wastewater is required to contain adequate amounts of Magnesium Mg^{2+} and Potassium K^{+} ions.
- The pH level should be close to neutral.
- The expected temperature range is between 10 to 25 degrees Celsius.

It is not recommended to use the model beyond the specified limitations.

2.7 Model Procedures for Activated Sludge Systems

To ensure accurate and dependable results, ASMs (Activated Sludge Models) need to be calibrated like any other model. However, calibrating ASMs can be a challenging task because they are complex environmental models. These models have multiple parameters that need to be evaluated, and several outputs that must be matched with actual data. Unfortunately, obtaining reliable data for quality control is often difficult because it requires significant financial and human resources. As a result, the models are typically over-parameterized, and identifying issues can arise (Mannina et al., 2011).

Various systematic calibration protocols have been proposed in recent years to support modelers during complex calibration studies. These include the STOWA protocol, BIOMATH, the WERF protocol, and HSG. (Mannina et al., 2011) and quite recently the GMP Unified Protocol (Rieger et al., 2013). These protocols have an important feature where they aim to estimate only a subset of parameters, rather than estimating all of them. The process of obtaining these subsets varies significantly across the different protocols (Mannina et al., 2011).

2.7.1 Steps in Modelling

The important step necessary in modeling the ASMs are as follows (Ghanesh, 2010);

1. Goals in modeling: The first step in the modeling process is to define the goal and identify the reason for modeling, expected outcomes, and limitations. A model is selected prior to data collection to understand the data requirements.
2. Data collection: Data is collected from the plant for calibration purposes, either through sampling or historical data. The amount of data collected depends on the model requirements, data reliability, and plant stability. Sampling is done when certain

parameters are not monitored on a daily basis. For steady-state model calibration, 24-hour composite samples are collected from major streams or in certain cases an average is taken of the plants data. A well-planned sampling program provides valuable information for reliable calibration.

3. Data Analysis: Performing data analysis is important to identify data that can be used during the model calibration stage. This is achieved by conducting mass balances and flow balances on input data, comparing the data collected during sampling with the historical data available at the plant, and conducting solids mass balances on clarifier data.
4. Model Calibration: Models consist of state variables, composite variables, and kinetic and stoichiometric constants. State variables are basic wastewater components like nitrates, ammonia, and biomass that are continuously integrated over time. GPS-X organizes state variables into different libraries. Common composite variables include TSS, VSS, COD, BOD, TP, soluble P compounds, TN, and TKN, and they are calculated from state variables. Initially, values for kinetic constants are assumed and then calibrated or modified until a good fit is obtained between the values predicted by the model and the actual measured value. During this stage, "sensitivity analysis" is performed, which measures the sensitivity of model output to changes in each of the model parameters. This is done systematically or based on the modeler's experience and knowledge. Sensitivity analysis is the most important step as it decides the reliability of a model.
5. Model Simulations: After calibrating the model, plant operators can perform simulations. These simulations involve testing the plant against "What if" scenarios. By doing so, operators can answer questions regarding responses to upgrades, plant problems during varying flow conditions, and more. Common simulations include feasibility assessments, aeration analysis, solids management, and biological nutrient removal.

2.8 Previous Works on Modeling Biological Nutrient Removal from WWTP

2.8.1 A Biological Treatment Plant Modeling Study of Kayseri WWTP (Topkaya, 2008)

The study utilized GPS-X, a simulation program, coupled with Activated Sludge Model No. 2d (ASM2d) to simulate nitrogen and phosphorus removal of the Kayseri wastewater treatment plant (WWTP). Calibration of the model employed data from March 2004, with validation occurring in May and July 2004. This selection allowed assessment under different climatic conditions, considering the potential impact on treatment efficiency (e.g., higher predicted nitrogen removal in summer).

Key findings from the study included:

- The model was successfully calibrated for most parameters except total phosphorus, potentially due to limited influent volatile fatty acid data.
- There was a good agreement between model predictions and measured values for underflow total suspended solids (primary clarifier), effluent mixed liquor suspended solids (aeration tanks), and underflow total suspended solids (secondary clarifier).
- Acceptable model performance for nitrogen components during calibration despite unexpected peaks in the simulation results.
- Stable predictions for chemical oxygen demand (COD) and total suspended solids, although daily fluctuations were not accurately captured.
- There were generally good validation results for the aforementioned parameters, with the exception of nitrogen components in the summer months (July 2004). Even with temperature adjustments, the model exhibited overestimation for these elements.
- The coefficient of determination (R^2) indicated limitations in goodness-of-fit assessment due to potential systematic bias in the data. The authors addressed this by employing the average ratio of predicted to measured values alongside standard deviation for a more robust evaluation.

2.8.2 The Modeling of the Biological Nutrient Removal Process at Gold Bar WWTP (Ghanesh, 2010)

The study aimed to calibrate a steady-state activated sludge model (ASM2d) for the BNR process at the Gold Bar WWTP (GBWWTP). The researcher collected historical data from the GBWWTP database, comprising 24 composite samples. Since experimental data for wastewater characterization was unavailable, a trial-and-error method was employed using the influent advisor module of the GPS-X software. The researcher then performed a sensitivity analysis to identify critical kinetic parameters, which they adjusted based on literature references. The complete BNR process was modeled using the ASM2d model within the Hydromatis GPS-X software.

Key Findings and Observations:

- Differences between the model-predicted and actual CBOD (carbonaceous biological oxygen demand) in the influent suggested discrepancies in wastewater characterization. Additionally, due to a lack of plant measurements, the exact orthophosphate concentration and nitrate/nitrite levels in the primary effluent were unknown. These unknown parameters were assumed for modeling purposes.
- Sensitivity analysis identified key parameters for calibration, including hydrolysis rate constants, anoxic hydrolysis reduction factors, and storage rate constants for polyhydroxyalkanoates (PHAs). The author suggested laboratory experiments to determine more accurate values for these parameters.
- Calibration yielded mixed results. While the mean squared error in CBOD prediction improved, the model still overestimated CBOD in the final effluent. Ammonia removal was generally overestimated, while nitrate/nitrite levels were underestimated. TSS (total suspended solids) remained largely unchanged after calibration, suggesting the need for separate secondary clarifier modeling for accurate TSS prediction. Orthophosphate prediction error also decreased significantly, but predicted values remained higher than actual measurements.
- The model indicated competition for CBOD between fast-growing heterotrophs and slower-growing phosphorus-accumulating organisms (PAOs). This resulted in a predicted higher denitrification rate and lower phosphorus removal compared to plant observations.

2.8.3 Modeling Biological Nutrient Removal in a Greywater Treatment System (Urdalen, 2015)

The study examined the use of activated sludge modeling for a new greywater treatment system, known as IFAS-EBNR-MBR (Integrated Fixed-Film Activated Sludge with Enhanced Biological Nutrient Removal and Membrane Bioreactor). The WEST software and ASM2d model were used to simulate the system. The study recognized the challenges of adapting the ASM2d model, which is typically used for conventional activated sludge systems, to the IFAS-EBNR-MBR configuration with membrane bioreactors (MBRs). To make the model applicable, several simplifications and assumptions were necessary.

Key Findings and Observations:

- The sensitivity analysis conducted identified six essential kinetic and stoichiometric parameters that have the most significant effect on the behavior of the model. To calibrate the model, an automatic parameter estimation algorithm (Simplex) was used, and for parameter variations, the ranges derived from the literature were utilized.

- The calibrated model accurately predicted the phosphorus distribution in the treatment tanks. However, it struggled to represent the primary anoxic P uptake process. The study suggests modifications to the ASM2d model to represent dominant pathways like this.
- A model was used to evaluate the sensitivity of a treatment system to operational parameters. Aeration and nitrified internal recycle were found to be the most sensitive parameters among all effluent quality variables. Scenario analysis was performed for different operational parameter combinations. An optimized parameter set was identified by balancing low total system volume with high P and N removal efficiency.

CHAPTER III: MATERIALS AND METHODS

3.1 Study Area

3.1.1 Presentation of Ain El Houtz in Tlemcen

The region being investigated is Ain El Houtz in Tlemcen, which is located in the northwestern part of Algeria, around 100 kilometers away from the Moroccan border. The Mediterranean Sea borders the region to the north, Naama city to the south, Morocco to the west, and the cities of Sidi-Bel-Abbes and Ain Temouchent to the east. The total area of the region is approximately 9,017.69 km^2 (RAHMOUN, 2019)

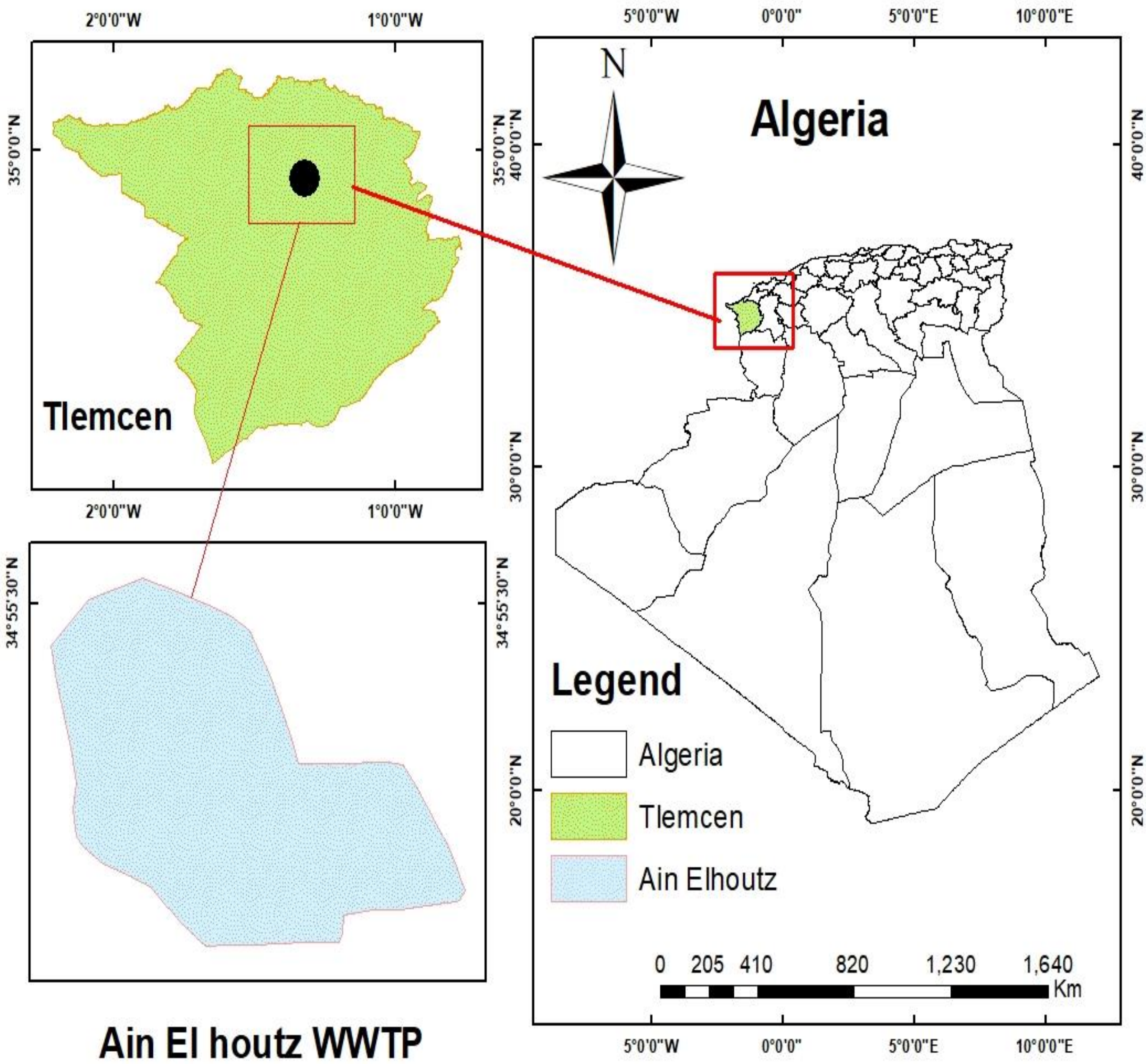


Figure 3. 1 The Map of the Study Area map

3.1.2 Wastewater Treatment Plant of Ain El Houtz (Tlemcen)

The Ain El Houtz wastewater treatment plant was established by the Office National de l'Assainissement (ONA), which is a public institution responsible for dealing with urban liquid waste pollution to protect the environment. It is located 6 km north of Tlemcen city, on the right bank of Ain El Houtz wadi, at the foot of Jebel Touma (Fig 3.2). The plant covers an area of approximately 17 hectares and has a treatment capacity of 30,000 cubic meters of wastewater per day. It was built by the Hydrotreating Company and began operating on November 5, 2005, to serve a population of 150,000. The Ain El Houtz WWTP plays a crucial role in enhancing the environmental sustainability of Tlemcen and the neighbouring areas as it extends to the communities of Abu Techfine and Oudjlida. The establishment of this plant serves various objectives, such as maintaining water quality in the Sekkak Dam, reusing treated wastewater for irrigation in the Hennaya perimeter, and utilizing sludge generated from the treatment for agricultural purposes, among others.

Figure 3. 2 Overview of Ain El Houtz WWTP

3.1.3 Description of Ain El Houtz WWTP treatment operations

The wastewater purification process at Ain El Houtz WWTP consists of 3 main stages: pre-treatment, primary treatment, and secondary treatment. Finally, we have the treatment of sludge. The figure below illustrates the operational mechanism of the WWTP (Fig 3.3).

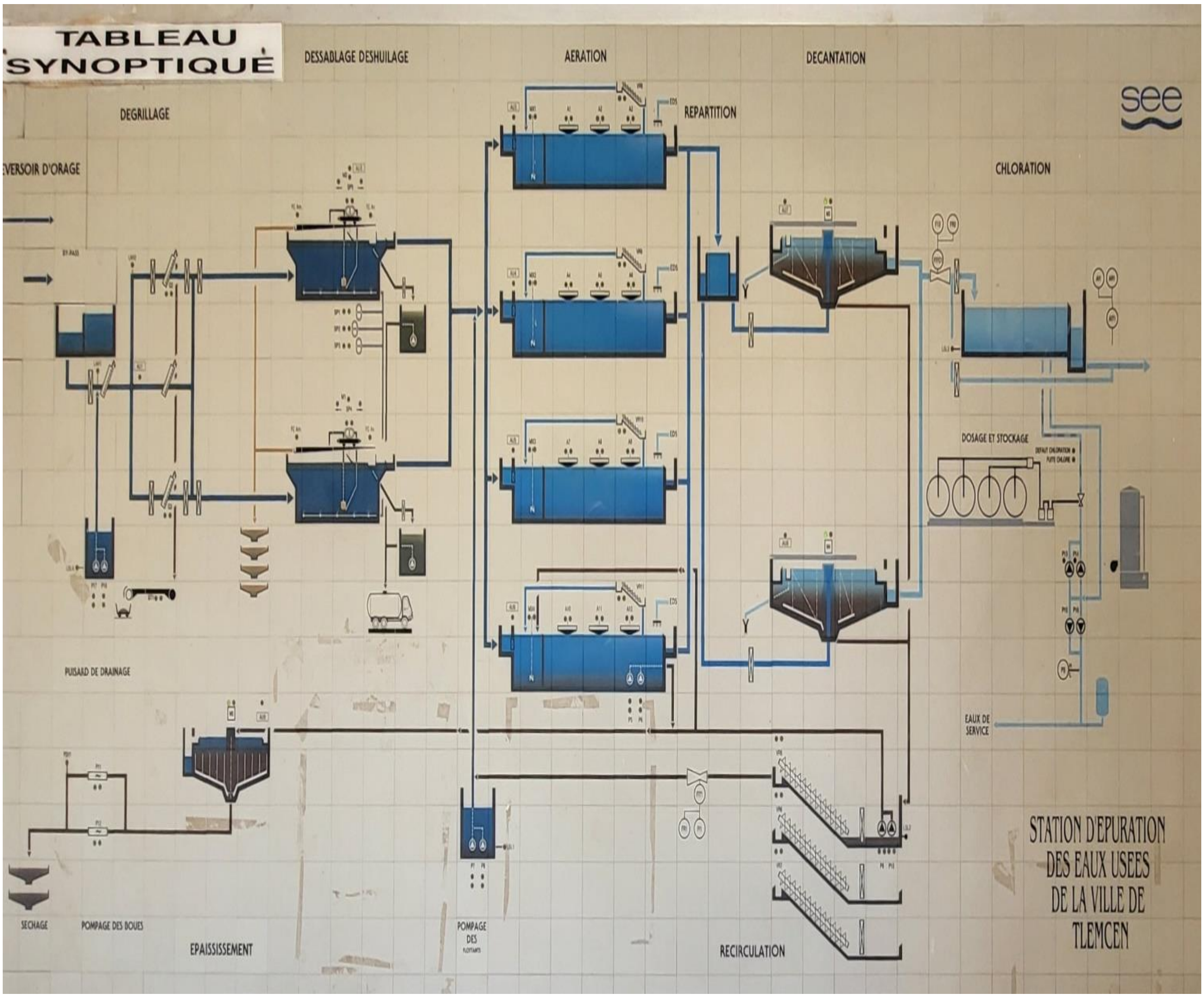


Figure 3. 3 Diagram representing the operational flow of Ain El Houtz WWTP

a) Primary Treatment

The purpose of the primary treatment stage is to remove the coarser elements that are likely to interfere with subsequent treatments and damage the equipment. These elements could damage the pumps, silt up the reactors, or interfere with the biological treatment (grease).

- **Screening**

Wastewater passes through a grid whose bars, more or less spaced apart, retain the most voluminous materials. The materials that are retained are conveyed through a conveyor belt to the refuse bin, where they are sent to either a landfill or incineration.

The WWTP includes two screens; the first is a manual screen, which comprises only one unit of coarse manual grid, inclined at 60^o with a grid width of 1.8m and spacing of 50mm. The screen is used to remove debris size above 50mm. The second screening process is a mechanized grid comprising 2 units with a grid width of 1.0m and a channel depth of 1.5m. The spacing between bars is 20mm with an inclination of 60^o. The motor power used at the screening chamber is 0.37kw

Figure 3. 4 Manual Coarse Screen

Figure 3. 5 Fine Screen

- **Grit Removal Chamber**

In this chamber, we have air blown into the chamber with the speed of the water reduced causing the sand to settle and collected at the bottom of the chamber. While the grease and scum or oil is collected at the top chamber, this is done to ensure the best performance of the aeration in the biological process. The Sand or grit and scum are taken out for treatment or disposal. For the Ain El Houtz WWTP, the grit chamber comprises two trapezoidal sand and oil separator channel; it is equipped with a suction bridge and skimmer 26 m long and 4 m wide.

Figure 3. 6 Grit Chamber

b) Secondary Treatment

- **Activated Sludge Basin**

The Ain El Houtz WWTP primarily treats wastewater through biological means, which involves the removal of dissolved and suspended organic matter through the activated sludge process. This process is mainly based on the activity of bacteria and other microorganisms. The primary objective of this process is to eliminate perceptible sensory effects and certain chemical elements present in the water. The biological treatment aims to eliminate carbon, nitrogen, and phosphorus pollution. This is achieved through the oxidizing and self-purifying capacities of microorganisms whose activity is enhanced by creating optimal conditions for them to degrade any organic and non-toxic pollutant.

The treatment plant uses a biological treatment process that involves four rectangular activated sludge tanks. Each tank has a volume of 4723 m3, a length of 55.5 m, a width of 18.5m, a water depth of 4.6 m, and a concrete height of 5.6 m. In each of the basins, three aerators provide oxygen to the microorganisms, which helps break down organic matter and stabilizes them. Each basin has an anaerobic zone that uses an internal recirculation process with an Archimedean screw to facilitate nitrification and denitrification.

Figure 3. 7 Activated Sludge Basin

Figure 3. 8 Internal Recirculation Process

The water after passing through all these steps then flows to the secondary clarifiers.

- ***Secondary Clarifier***

After the biological reaction, the mixed liquor from the aeration tanks goes to the clarifiers or settling tanks. Here, a final decantation process occurs which separates the purified water from the bacterial flocs, also known as sludge. The clarifier is a critical component of the biological wastewater treatment process. The purified water then moves to the next stage of the treatment process, while the sludge moves to the thickener for further treatment.

The treatment plant consists of two circular settling tanks, each with a diameter of 46 meters. The surface area of each tank is 1661 cubic meters, with a water depth of 4 meters at the periphery. Each tank is equipped with a scraper bridge that has a rotation speed of 0.04 meters per second.

Figure 3. 9 Secondary Clarifier

c) Tertiary Treatment

- **The Disinfection Basin**

The wastewater treatment plant of Ain El Houtz is equipped with a disinfection basin with the The total volume is estimated to 700 m^3. The installed dosing capacity and 40 Kg of chlorine per hour. The reagent used for disinfection is the sodium hypochlorite (NaCLO). The expected contact time is 43 minutes for the average daily inflow rate and 14 minutes for peak inflow rate during rainy period. However, the chlorine disinfection treatment is suspended in the WWTP.

Figure 3. 10 Disinfection Basin

Then, the treated wastewater will flow through a canal to be discharged and used for irrigation in most cases

Figure 3. 11 Outlet from the WWTP

d) Description of the Treatment Processes of the Sludge

The treatment processes of the sludge are comprised of the following steps:

- **Recycling and Disposal of the Exceeded Sludge**

During this step, the activated sludge is subtracted from the bottom of the clarifier, returned back to the top of the biological treatment, in order to regenerate it and to maintain a substantially constant concentration in purifying microorganisms. A submerged pump and another one in reserve, placed in a well, provide this recycling operation. Each pump have the following characteristics: The unitary flowrate is equal to 455 m^3/hr and the manometric height is equal to 5.0 m.

In the same well, there are two other pumps, of which one in reserve, for the evacuation of exceeded sludge towards the thickener. The characteristics of these pumps are: The flow rate is equal to 30 m^3/hr, the total manometric height is equal to 5.0 m. The maximum concentration of sludge extracted from the clarifier is approximately 0.8%.

In each recycling line, a flow meter is placed to allow the possibility to adapt the recycling flowrate according to the flowrate of the wastewater entering in the WWTP.

Figure 3. 12 Sludge Pumping Station

- **Thickening of Biological and Chemical Sludge**

This treatment is the first stage of a significant reduction of the volume of the sludge from the biological and chemical treatment of the wastewater. The exceeded sludge is directed to a circular thickener with a diameter of 14 m, and the total height is 4m and a bottom slope of 1/10. Mechanization is applied in a scraping and slow stirring system to facilitate the sludge slippage to the central pit from which they are extracted and to allow the release of interstitial water and gases contained in the sludge.

Figure 3. 13 Sludge Thickener

- ***Drying Beds***

The thickening mud is taken by pumping and discharged to the drying beds. Sludge drying is carried out in open air in areas of areas 30 m long and 15 m wide. There are 14 drying beds made of concrete equipped with a perforated drainage pipe, to allow the evacuation of filtered water to the entrance of the station. The drying time in the plant is normally about 4-6 weeks, however, it can reach 3 to 4 months during unfavorable weather conditions.

Figure 3. 14 Drying Beds

3.2 Collection and Evaluation of Data of Ain El Houtz WWTP

To gather the necessary details about the influent and effluent characteristics of Ain El Houtz WWTP, we reviewed the physical data of each process unit, as well as laboratory data analysis and daily operation bulletins. Collection and Evaluation of Data of Ain El Houtz WWTP. We obtained this data from the Office National de l'Assainissement (ONA), the organization responsible for the information. The physical parameter analysis, including turbidity, pH, conductivity, temperature, and dissolved oxygen (DO), were conducted daily. However, the chemical oxygen demand (COD), biochemical oxygen demand (BOD_5), total suspended solids (TSS), ammonium ion (NH_4-N), nitrite (NO^-_2-N), nitrate (NO_3^--N), and orthophosphate (PO_4^{-3}) were analyzed once a week.

The descriptive statistics, including mean, median, standard deviation (SD), minimum (Min), maximum (Max), and removal efficiency (RE), were used to evaluate the important parameters of the influent and effluent samples of the Ain El Houtz WWTP.

3.3 Model Development and Calibration

This research employed the GPS-X software version 8.5 that developed by Hydromantis Environmental Software Solutions, Inc. based in Ontario, Canada. This is an advanced software tool that enables mathematical modeling, simulation, optimization, and management of wastewater treatment plants. Its user-friendly interface includes a comprehensive selection of unit processes and a simple drag-and-drop feature that allows users to easily create a plant model, input characterization data, and run simulations (GPS-X Technical Reference, 2022). The GPS-X simulator functions by conducting a material balance over every state variable in the ASM model throughout the process units. This includes the flow rates entering and exiting the process units, as well as the designated generation or consumption rate. The software offers diverse techniques for inputting COD, nitrogen, phosphorus, and solids fractions of influent state variables (Tiar et al., 2024).

3.3.1 Building the Ain El Houtz WWTP Model GPS-X

The Ain El Houtz WWTP was constructed using the GPS-X modeling platform. To achieve this, process unit icons from the process table of GPS-X's unit process library were selected and connected with flow paths to build the process flow diagram of the WWTP, as shown in the Fig below.

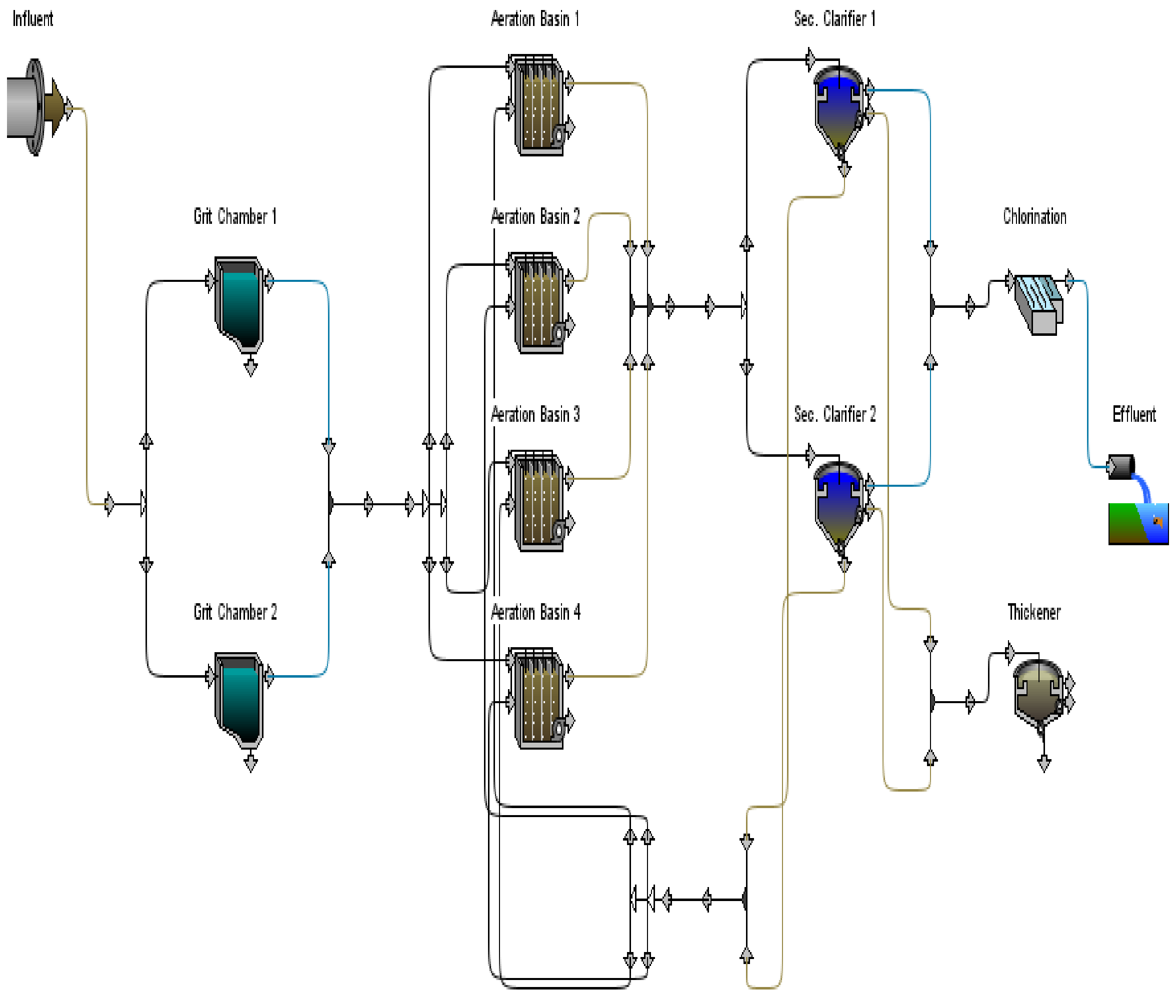


Figure 3. 15 Layout of the Ain El Houtz WWTP on GPS-X

3.3.2 Selection of the library

In order to begin characterizing each unit process, it was necessary first to select the appropriate library for Biological Nutrient Removal at the WWTP plant, and to choose a library that includes the ASM2d model. A library in GPS-X refers to a collection of wastewater process models that use a set of basic wastewater components or state variables. These state variables are continuously integrated over time. GPS-X offers nine libraries with default values and expressions for calculating state variables. As the focus of this research was on modeling the BNR process of the WWTP, the ASM2d model was chosen from the Carbon, Nitrogen, Phosphorus (CNPLIB) library as it includes biological phosphorus removal and simultaneous nitrification-denitrification (GPS-X Technical Reference, 2022).

3.3.3 Selection of the model for each process unit

For every process unit defined, there is a set of models available to describe the behavior of the object. The model's choice depends on the information available to fill the user inputs necessary for calculating state and composite variables.

- **Influent**: The influent wastewater's characteristics are the foundation of the simulated system because they impact the rest of the WWTP's behaviour. GPS-X provides six models for influent characterization, including bodbased, codfractions, codstates, sludge, states, and tsscod. The codstates model was selected for influent characterization. Using this model, most of the state variables were calculated as a fraction of the total COD, with default values already provided in the influent advisor of the software. However, the user can modify these COD fractions to calibrate the WWTP plant model better.

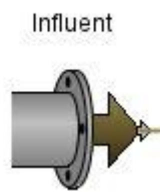
- **Aeration Tank**: ASM2d was the model selected to describe the biological treatment in the aeration tank.

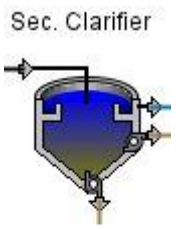
- **Clarifiers**: For the modelling of the secondary clarifiers, simple1d was selected

- **Grit Chamber**, **Thickener, and chlorination**: The selected model for modelling these unit processes was the empiric model, which is the default model.

3.3.4 Physical and Operational Data

For the calibration of the physical and operational data of Ain El Houtz WWTP, data of each process unit collected from the Office National de l'Assainissement (ONA) was used as shown in the table below:

Process Unit	Physical Parameter	Value	Operational Parameter	Value
			Flow Data	30000 m^3/d

	Number of tanks	4		
	Tanks Depth	4.6 m		
	Max. Volume	4723m^3		
	Number of reactors (in each tank)	4		
	Clarifier Type	Flat Bottom	Pumped flow	120 m^3/d
	Surface	1661 m^2		
	Water depth	4 m		

3.3.5 Influent Characterization and Model Calibration

Once we have entered the relevant physical and operational parameters and chosen the appropriate library, we proceed to analyze the influent using the advanced influent advisor feature within the GPS-X software. Basic influent wastewater metrics, including BOD_5, BODu, COD, TSS, VSS, and TKN, are critical to establishing accurate mass balances throughout the system. By examining the influent's suspended solids, VSS, and BOD levels, we can determine the various organic fractions present in the influent (GPS-X Technical Reference, 2022).

To calibrate the model, the initial step involves calibrating it to a steady state. This entails taking the data from the Ain El Houtz WWTP and averaging it under the assumption that this average represents a constant state. The model is then fine-tuned to match the average effluent data (Makinia, 2010). For the calibration of Ain El Houtz WWTP, the average was taken for two years (2020 and 2021). The table below represents the data used in the calibration of the WWTP

Table 3. 1 Influent and Calibrated value for calibration process of Ain El Houtz WWTP

S/N	Parameters	Influent value	Calibrated Value
1	COD	375	375
2	BOD_5	194	194.5
3	TSS	222	223.7
4	NH_4	40	40
5	PO_4^{-3}	8	8
6	NO_3^- & NO_2^-	4	4
7	TKN		45
8	TP		10
9	DO		2

To achieve the TSS and BOD_5, some parameters were corrected to achieve maximum match between the model and influent data. Table 3.2 shows the corrected parameters in the model

Table 3. 2 Calibrated value of the Influent Advisor on GPS-X

Symbol	**Description**	**Unit**	**Default Value**	**Calibrated Value**
ivt	VSS/TSS ratio	gVSS/gTSS	0.75	0.68
frsi	Soluble inert fraction of total COD		0.05	0.07
frxi	Particulate inert fraction of total COD		0.13	0.145

3.4 Model Validation

Once the plant had been successfully calibrated, the next task was to carry out validation. Model validation refers to the model's predictions being in excellent agreement with a different dataset that was not used during the model's development, within acceptable limits (Mu'azu et al., 2020). This is done to test the model's predictability for forecasting other possible events or scenarios related to the treatment plant.

To validate the model, we perform a dynamic modeling of the wastewater treatment plant (WWTP) data from the year 2022 and assess the error through two criteria: the root mean squared error (RMSE) and the mean absolute error. RMSE serves as the maximum likelihood estimator of the variance of model predictions under normal distribution, emphasizing higher errors and indicating the average magnitude of errors without compensation. The goal is to minimize RMSE. On the other hand, the mean absolute error gauges variability, and the objective is to minimize this criterion as well (Rieger et al., 2013).

$$RMSE = \sqrt{\frac{1}{n}\sum_{i=1}^{n}(r_i)^2} \quad (3.1)$$

$$MAE = \frac{1}{n}\sum_{i=1}^{n}|r_i| \quad (3.2)$$

r_i = Residual = O_i - P_i

O_i = Observed values

P_i = Predicted values

CHAPTER IV: RESULTS AND DISCUSSION

4.0 Introduction

This chapter presents the findings and discussions on the modeling of biological nutrient removal (BNR) for the Ain El Houtz WWTP. It covers the influent and effluent wastewater characterization and an analysis of the processes involved in developing the model. The simulation results from the dynamic model were then compared with the plant results to validate the model.

4.1 Wastewater Characterization

To evaluate the efficiency of Ain El Houtz wastewater treatment plant (WWTP), the physicochemical parameters for the influent and effluent wastewater data were processed by the laboratory and the data for the years 2020 to 2022 were used. The plant's weekly laboratory data was analyzed to identify temporal changes in water quality of influent and effluent water. The 128 samples were collected after removing outlier's data. The collected data was then statistically analyzed to assess the WWTP's ability to treat and remove pollutants such as BOD_5, COD, TSS, PO_4^{3-}, NO_2^-, NO_3^-, NH_4, and pH parameters. The average of the three years was used to calculate the removal efficiency of the WWTP. Descriptive statistics are presented in Table 4.1 and Table 4.2, including the mean, median, standard deviation (SD), minimum (Min), maximum (Max), and removal efficiency (RE). the Key observations are provided below.

- Total Suspended Solids (TSS): Both the influent and effluent exhibit significant TSS levels, indicating the presence of organic and inorganic particulate matter. The high removal efficiency (RE) of 91.2% suggests that the primary and secondary treatment processes are effective. A study conducted by Rahmoun (2019) showed similar results, which were below the limit value set by the Algerian government for treated wastewater for irrigation (30mg/l).

- Chemical Oxygen Demand (COD): Similar to TSS, both the influent and effluent COD values are high, indicating a high organic matter load. The impressive RE of 90% confirms the efficient removal of organic pollutants. The upper limit (79 mg/l) of COD values in the effluent from Ain El Hout WWTP (Table 4.2) was different from values obtained by Medjda (2021). He found that the upper limit for COD values was 41 mg/l, which is quite below the limit of the standard value set by the Algerian government for water intended for irrigation (90 mg/l).

- Biochemical Oxygen Demand (BOD_5): The influent BOD_5 highlights a substantial biological oxygen demand, indicating high organic matter readily biodegradable by microorganisms. The RE of 92.2% suggests that the biological treatment processes are efficient. However, we noticed a peak in the concentrations of influent BOD over a few days, with a maximum of 1150 mg/l. This indicates a high content of easily degradable organic material in the wastewater (Rahmoun, 2019). This peak also affects the concentration of the effluent, as we noticed that the maximum of 35mg/l is above the Algerian standard (30mg/l) for irrigation (Medjda, 2021).

- Ammonium- Nitrogen (NH_4): The influent NH_4 levels are relatively moderate, which may be due to nitrification occurring before entering the plant. The moderate removal efficiency (RE) of 60% indicates that there may be partial nitrification/ammonification processes or incomplete nitrification, with the influent values ranging from 8-102 mg/l and the effluent values ranging from 0-39 mg/l. During a study conducted by Rahmoun (2019), it was observed that the maximum seasonal value of ammonium-nitrogen concentrations for treated wastewater from the Ain El Houtz WWTP was recorded during the summer season, while the minimum value was recorded during the spring season. The ammonium concentration recorded during this study period was below the FAO's limited value set for irrigation water (5 mg/l).

- Nitrite- Nitrogen ($N\text{-}NO_2^-$): The concentrations of $N\text{-}NO_2^-$ in both the influent and effluent were found to be low, indicating rapid conversion to nitrate via nitrification. The influent concentrations varied between 0 mg/l and 4 mg/l, while the effluent concentrations were between 0 mg/l and 2 mg/l. Medjda (2021) also recorded similar values for nitrite in the treated wastewater from the Ain El Houtz WWTP, which varied between 0.43 mg/l and 1.8 mg/l before treatment, and between 0.29 mg/l and 0.9 mg/l after treatment. The nitrite variation in the treated water was below the limit set by the Algerian government for irrigation water, which is 1 mg/l.

- Nitrate- Nitrogen ($N\text{-}NO_3^-$): The levels of $N\text{-}NO_3^-$ in the influent and effluent indicated efficient nitrification and potential denitrification, with values ranging from 0 to 17 mg/l and from 0 to 13 mg/l for the influent and effluent discharge, respectively, which is within the limit set by the Algerian government (30 mg/l). According to a study conducted by Medjda (2021), the nitrate-nitrogen levels in untreated wastewater ranged between 2.4 and 3.6 mg/l, whereas after treatment, the levels varied between 1.2 mg/l and 1.8 mg/l for the same WWTP. This can be attributed to the effectiveness of nitrification-denitrification bacteria during the biological treatment process. The increase in nitrogen load from the municipality might have caused the difference in values, requiring longer hydraulic retention time for nitrification during that period.

- Phosphate (PO_4^{-3}): Moderate levels of phosphate ions in the influent and effluent suggest the presence of inorganic phosphorus. Rahmoun (2019) recorded that the phosphorous values were high (4.12 mg/L), which surpasses the limit for standards set by the FAO and the Algerian government for irrigation water (2 mg/L). It was likely due to extensive discharge from sewage and surrounding agricultural areas during spring season, and possibly a malfunction of the wastewater treatment plant. However, the laboratory data we obtained for this study showed even higher values in the range of 28 mg/L during peak times, which is similar to the influent values. This indicates the need to implement a process for phosphorus removal in the treatment plant.

Table 4. 1 Influent Characteristics

Parameters	Unit	Mean	Min	Max	Median	S.D	Removal Efficiency (R.E)
TSS	mg/L	232	103	848	230	72.84887	91.19698
COD	mgCOD/L	381	120	1324	345	156.4514	90.01149
BOD_5	mgO2/L	202	0	1150	180	119.3857	92.22795
NH_4	mgN/L	41	8	102	40	12.59732	59.93358
$N\text{-}NO_2$	mgN/L	1	0	4	1	0.670134	39.47901
$N\text{-}NO_3$	mgN/L	5	0	17	3	4.528693	28.66946
PO^{3-}_4	mg/L	8	2	28	7	3.639338	34.85062

Table 4. 2 Effluent Characteristics

Parameters	Unit	Mean	Min	Max	Median	S.D	Removal Efficiency (R.E)
TSS	mg/L	20	11	42	19	5.090562	91.19698
COD	mgCOD/L	38	16	79	36	13.07997	90.01149
BOD_5	mgO2/L	16	0	35	15	6.10458	92.22795
NH_4	mgN/L	16	0	39	14	9.066463	59.93358
$N\text{-}NO_2$	mgN/L	1	0	2	1	0.383155	39.47901
$N\text{-}NO_3$	mgN/L	4	0	13	2	3.830541	28.66946
PO^{3-}_4	mg/L	5	0	28	5	3.323772	34.85062

4.2 Model Calibration

As discussed in Chapter 3, The GPS-X software has several libraries that provide corresponding kinetic, stoichiometric, and wastewater fractions with default values. However, when the default wastewater fractions do not give the needed values, modifying them to calibrate the model becomes necessary. After inputting the necessary calibration values as outlined in chapter 3 (part 3.3.5), and selecting the ASM2d model for the biological nutrient removal (BNR) process, sensitivity analysis was carried out on the kinetic and stoichiometric parameters to determine which ones would be sensitive to the model. This was done using the step analyze function in the analyze mode of the GPS-X software. As mentioned before, the objective of this study was to identify the parameters that would have a significant impact on the model's effluent. For each kinetic and stoichiometric parameter, the numerical value was increased, a graph was plotted for each variable, and a simulation was run to ensure a constant trend. From the analysis, it was observed that the stoichiometric parameters did not have a significant impact on the effluent, while a few of the kinetic parameters showed high sensitivity to the effluent values.

In a similar study conducted by Topkaya (2008), GPS-X was used to model Kayseri WWTP. The study identified independent variables such as BOD, COD, etc., and performed sensitivity analysis on these sensitive parameters, affecting them during the modeling process. During sensitivity testing, the minimum, maximum, and increment values were entered for the independent variable. GPS-X then displayed various output graphs showing parameter values versus selected output parameters. According to the output graph, the sensitive parameters were then used in the calibration of the model. This study identified and calibrated 7 parameters based on the sensitivity analysis.

4.3 Steady-state simulation results

In this research, the primary goal of steady-state model calibration is to match the simulated values of each variable with the mean values obtained from the plant. This approach has been commonly used in many studies to evaluate the performance of Activated Sludge Systems in large scale similar to Ain El Houtz WWTP (Tiar et al., 2024)

After entering necessary data such as flow rate, COD, BOD, TSS, etc., the simulation was run. During sensitivity analysis, it was noticed that the phosphorus content was insensitive or quite slightly sensitive to both kinetic and stoichiometric parameters. Topkaya (2008) noticed a similar trend in his study, where the effluent phosphorus concentration was insensitive to kinetic parameters related to phosphorous kinetics in the selector and aeration tanks. Based on these observations, the phosphorus content was not calibrated

Regarding the nitrogen content of the model, which includes ammonium, nitrate, and nitrite ions, it was observed that certain kinetic and stoichiometric parameters were sensitive to calibration. However, the degree of sensitivity varied between parameters. Therefore, we calibrated six parameters that have been widely used in previous literature.

The reduction factor for denitrification [η_{NO3}] for heterotrophic organisms was increased from 0.8 to 3, which is significantly higher than the values recorded in different literatures. According to Topkaya (2008), the range of values for the reduction factor for denitrification was between 0.6 to 1. Although the values used in this study were quite higher than the recorded value, it was necessary to assume the occurrence of a significantly greater denitrification reduction factor to improve the fitting of the NO_3-N and NO_2-N simulation curve and maintain the fitting for the NH_4-N simulation curve.

The maximum specific growth rate for autotrophic biomass was reduced slightly from 1 d^{-1} to 0.985 d^{-1}, which is still within the acceptable range reported by various literatures. The value widely reported was 0.2 d^{-1} to 1.2 d^{-1} (Chen et al., 2020; Urdalen, 2015; Z. Wang et al., 2022). Insel et al. (2003) reported a slightly different range of 0.25 d^{-1} to 1.23 d^{-1}. The saturation coefficient for ammonium (substrate) [K_{NH4}], it was increased from 1 g N m^{-3} to 1.4 g N m^{-3}

which was the same value reported by Makinia et al. (2006). However, it was slightly higher than the value of (1.3 g N m^{-3}) that reported in the study conducted by Drewnowski & Makinia (2013). Moreover, the range of values reported in literature is between 0.5 g N m^{-3} to 1.5 g N m^{-3} (Topkaya, 2008).

The oxygen saturation coefficients for heterotrophic [Ko_{2H}] and autotrophic [Ko_{2A}] biomass was increased from 0.2 g O_2 m^{-3} to 0.5 g O_2 m^{-3} and from 0.6 g O_2 m^{-3} to 0.79 g O_2 m^{-3}, respectively. This change was made to account for oxygen diffusion limitations in the system during simultaneous nitrification and denitrification. It was hypothesized that both coefficients needed to be changed, not just one of them (Larrea et al., 2002). Previous studies have reported ranges for the oxygen saturation coefficients for heterotrophic [Ko_{2H}] and autotrophic [Ko_{2A}] biomass: 0 g O_2 m^{-3} to 0.5 g O_2 m^{-3} and 0.2 g O_2 m^{-3} to 0.8 g O_2 m^{-3}, respectively (Topkaya, 2008); and 0.1 g O_2 m^{-3} to 1 g O_2 m^{-3} and 0.1 g O_2 m^{-3} to 2 g O_2 m^{-3}, respectively (Urdalen, 2015). Although the ranges are slightly different, the calibration of the Ko_{2H} and Ko_{2A} coefficients falls within the ranges reported in both studies.

The autotrophic organisms' decay rate (b_{AUT}) was increased from 0.15 d^{-1} to 0.5 d^{-1}, which is considered an acceptable range according to the literature, which falls between 0.05 d^{-1} to 2 d^{-1} (Wang et al., 2022). The summarized value for the default and calibrated values for the six parameters was tabulated in Table 4.3.

Table 4. 3 lists the parameters calibrated for this study's model.

S/N	Kinetic Parameter	Unit	Default Value	Calibrated Value
	Heterotrophic organisms: X_H			
1	Reduction factor for denitrification [η_{NO3}]		0.8	3
2	Saturation/ inhibition coefficient for oxygen (or Oxygen half-saturation coefficient (on GPS-X)) [K_{o2H}]	g O_2 m^{-3}	0.2	0.5
	Nitrifying organisms (autotrophic organisms):X_{AUT}			
3	Maximum growth rate of autotrophic organisms [μ_{AUT}]	d^{-1}	1	0.985
4	Decay rate of autotrophic organisms [b_{AUT}]	d^{-1}	0.15	0.5
5	Saturation coefficient for ammonium (substrate) [K_{NH4}]	g N m^{-3}	1	1.4
6	Saturation/ inhibition coefficient for oxygen X_{AUT} (or Oxygen half-saturation coefficient (on GPS-X)) [K_{o2A}]	g O_2 m^{-3}	0.6	0.79

In order to evaluate the performance of a model in a steady-state mode, we use statistical measures such as Mean Absolute Error (MAE) and Root Mean Squared Error (RMSE). The MAE is the average of the absolute errors between the simulated and observed values. The MAE of zero value indicated that the mathematical model can accurately reproduce the physical one, and the value increases as the discrepancies grow. On the other hand, RMSE is useful for determining the typical magnitude of model errors (Cosenza et al., 2009).

In the steady state model, it was observed that the MAE and RMSE were the same for NH_4, which is 6.06. This indicates a significant difference between the simulated and observed values for ammonium concentrations. The simulated value for nitrite and nitrate was higher than the observed value. RMSE and MAE values for this error were 1.36, indicating a moderate level of error in the simulation. Similar to NH_4, there is a discrepancy between the simulated and observed concentrations for phosphate, with the simulated value being higher than the observed value. The RMSE and MAE values for this error are both 3.167, that indicating a substantial error level in the simulation. Table 4.4 shows the steady-state values for MAE and RMSE for the model

Table 4. 4 Steady-state value for MAE and RMSE

S/N	Variable	Simulated	Observed	RMSE	MAE
1	NH_4	23.06	17	6.06	6.06
2	NO^{2-} & NO^{3-}	4.36	3	1.36	1.36
3	PO^{3-}	9.167	6	3.167	3.167

Figures 4.1 to 4.3 present the results of a calibrated steady-state model simulation of the effluent concentrations of the biological nutrients: NH_4 (Ammonium), PO_3^- (Phosphate), and NO_2^- (Nitrite) & NO_3^- (Nitrate) in the Ain El Houtz WWTP.

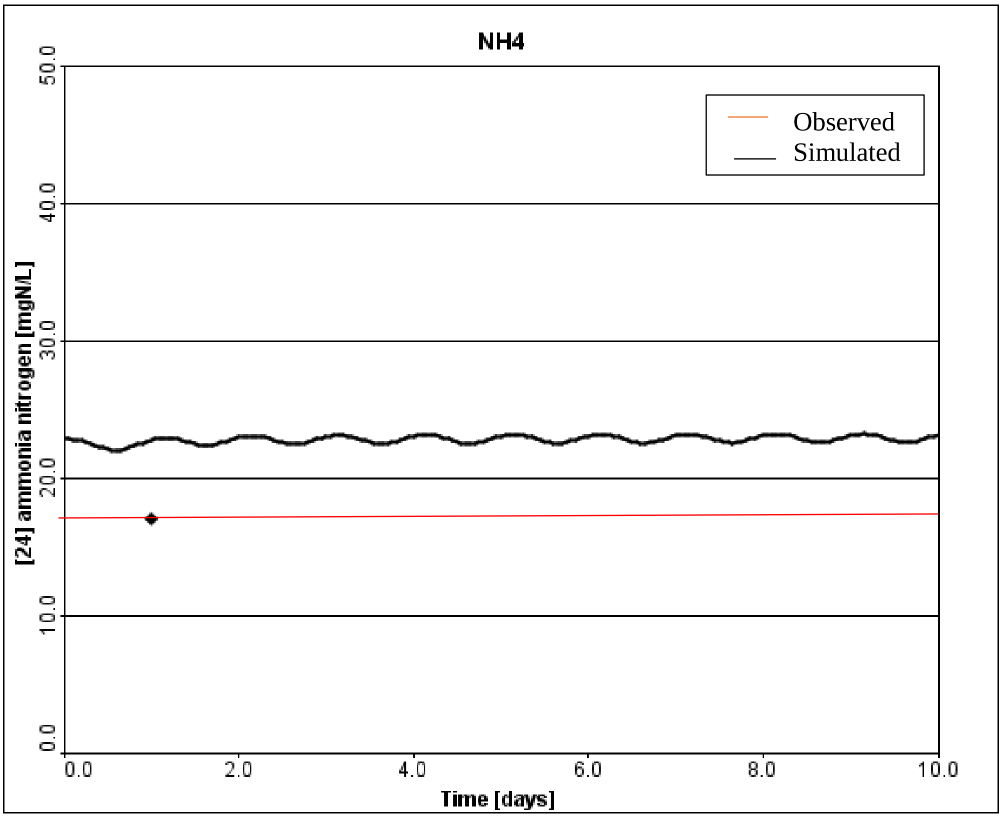


Figure 4. 1 NH_4 Steady state calibration results

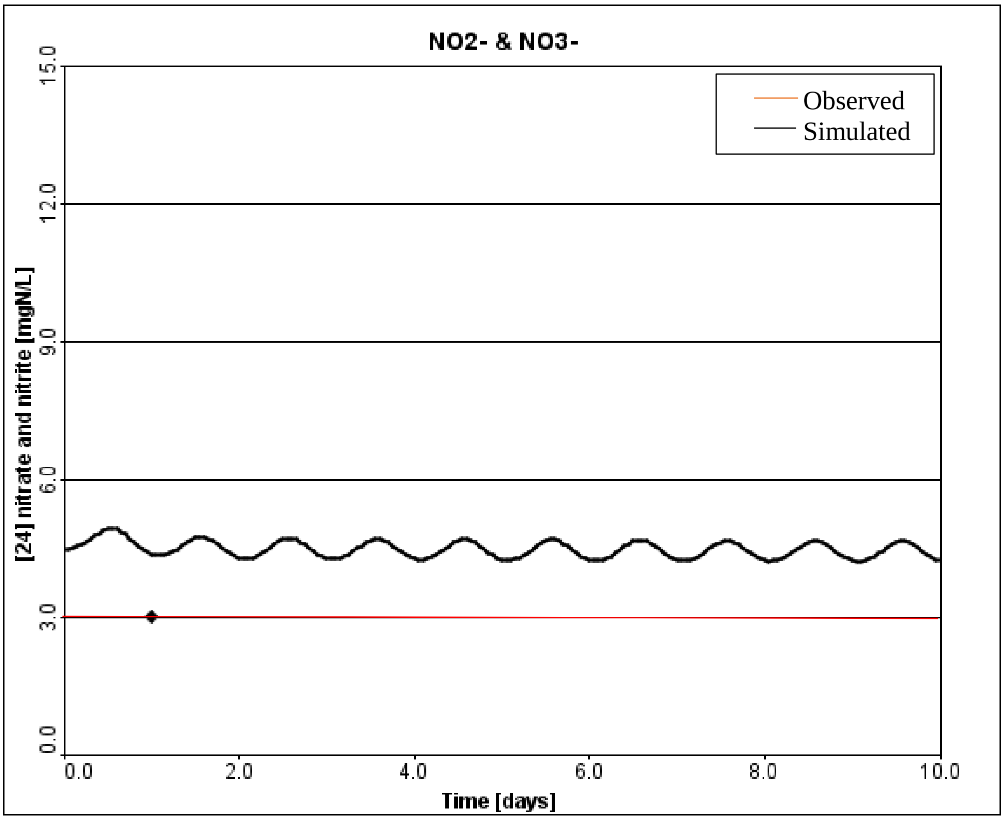


Figure 4. 2 NO_2^- & NO_3^- Steady State calibration results

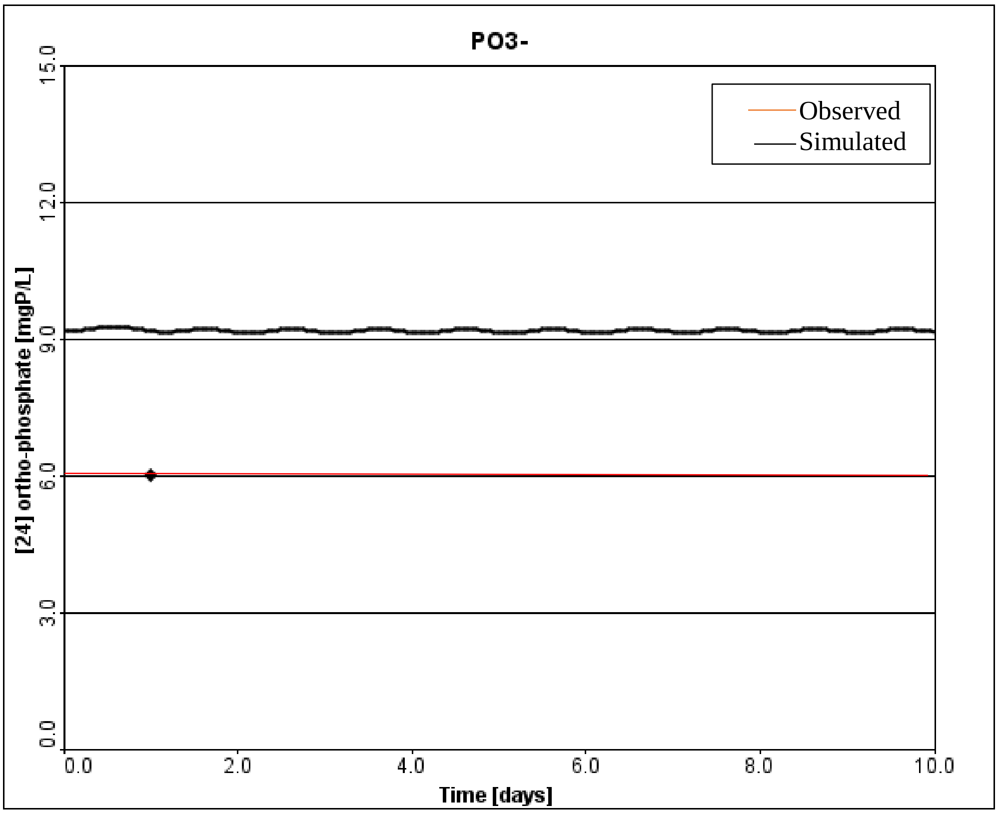


Figure 4. 3 PO_3^- Steady State calibration results

4.4 Dynamic Model

After the calibration process was completed, we evaluated the performance of the model through dynamic validation. This process included using data from a one-year period (2022) that consisted of 30 individual data points. In Rathore's (2018) study, they also used a 30-day period of dynamic data during the validation of the model. The validation process employed the calibrated model within the influent controller, which simulated the system's behavior using the data. Specifically, the controller received influent data points encompassing various concentrations, including Chemical Oxygen Demand (COD), flow rate, Ammonium ion (NH_4), Phosphate ions (PO_3^-), and nitrite & nitrate (NO_2^- & NO_3^-) ions. The validated model was then used to simulate the system's response for a period of 30 days, employing the same calibrated parameters as in the steady-state analysis.

Figures 4.4 to 4.6 illustrate the results of the simulation, showing graphical representations of the simulated effluent concentrations as compared to the actual effluent data. These visualizations provide a comprehensive overview of the model performance in simulating the effluent concentrations of the BNR over the 30-day simulation period.

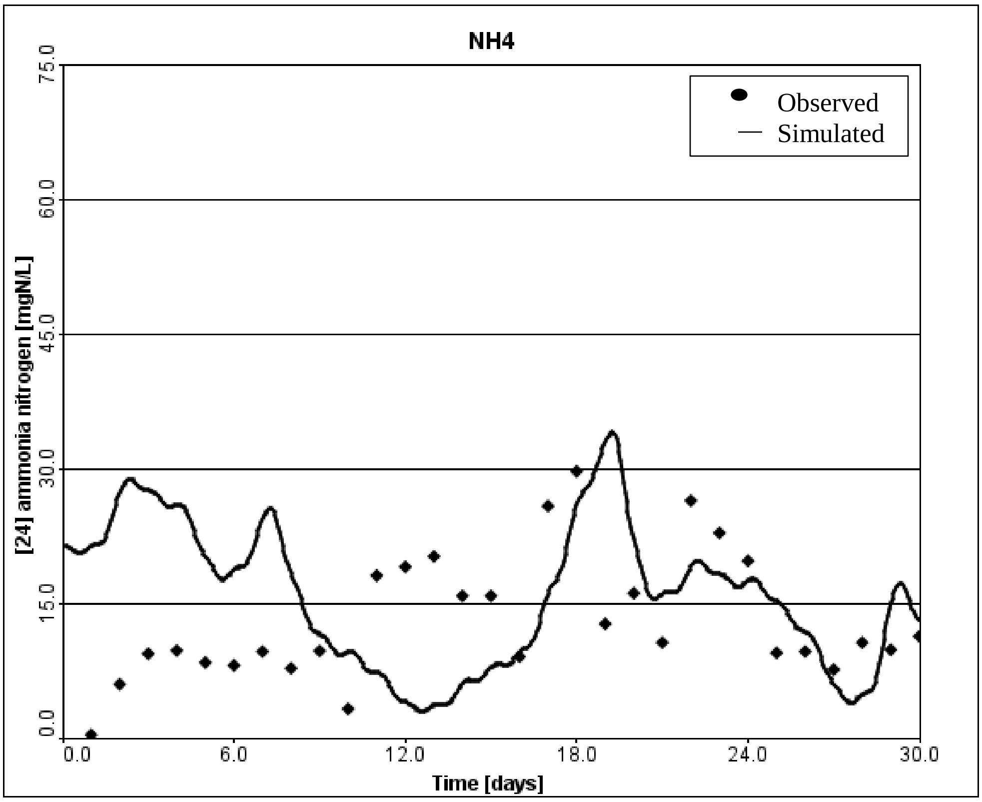


Figure 4. 4 NH_4 Dynamic State validation results

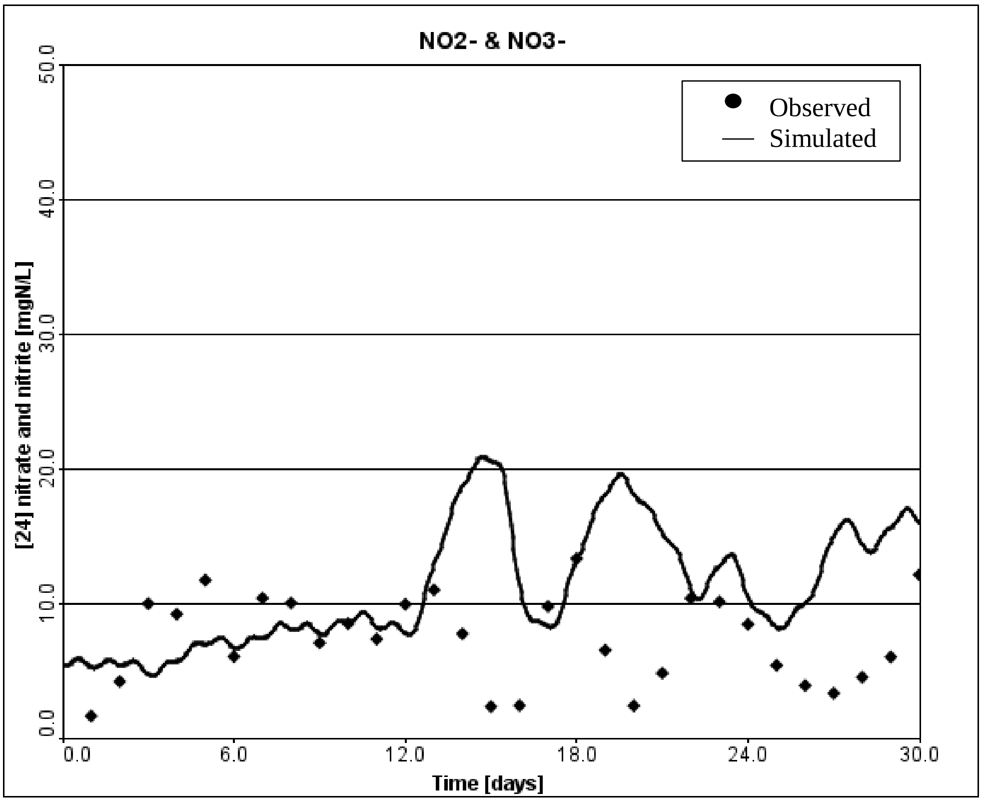


Figure 4. 5 NO^{2-} & NO^{3-} Dynamic state validation results

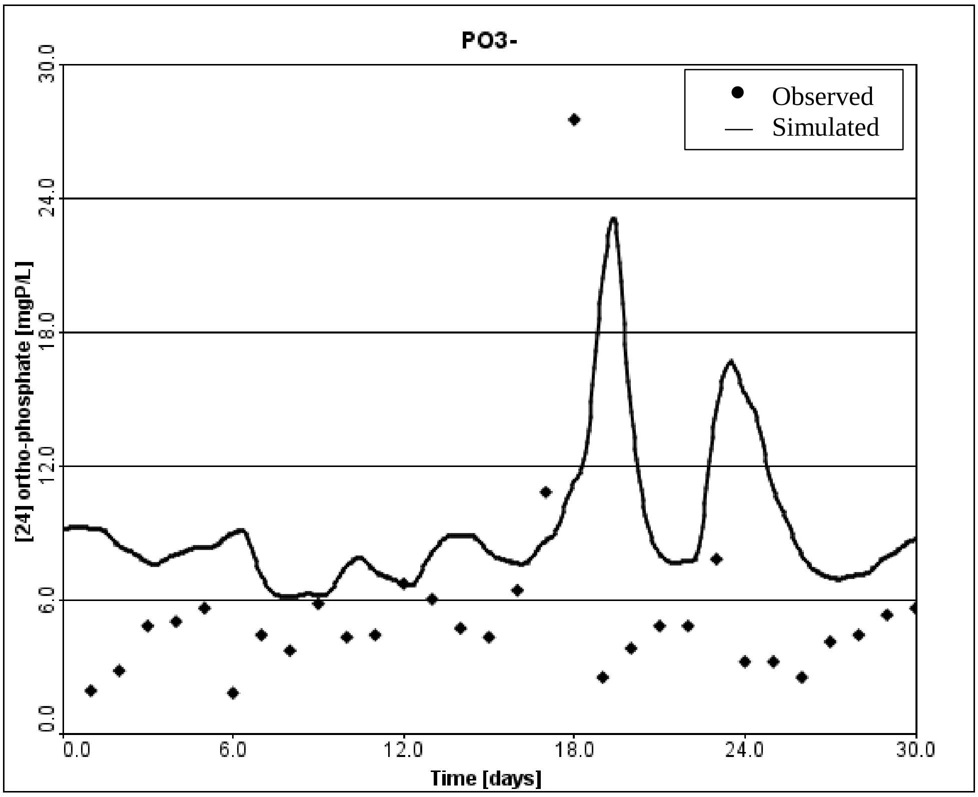


Figure 4. 6 PO^{3-} Dynamic state validation results

In the dynamic state model, it was observed that there were discrepancies in the NH_4 model with the model underestimating and overestimating the effluent concentration. Additionally, different peaks were observed, indicating that the influent at that point was a bit high. Rathore (2018) reported a similar trend in the simulation of a 30-day dynamic model, which was attributed to the high influent concentration recorded in the plant data on those specific days. This is further supported by the values recorded for the RMSE and MAE for NH_4, which were 13.40 and 10.14, respectively (Table 4.5).

The model for the nitrite and nitrate concentrations also noticed the similar trend of the model overestimating and underestimating the effluent concentrations. This suggests that the different rate of denitrification in the model and quite different from that of the reality, which may have been due to the variation of influent concentrations at Ain El Houtz WWTP. Ghanesh (2010) reported that the nitrate concentration was underestimated even after calibration, attributing it to the BNR design of the WWTP. In this design, the influent stream rich in biodegradable organics first comes into contact with the anoxic zone, followed by the anaerobic zone, where heterotrophs and PAOs consume most of the CBOD. Heterotrophs grow faster and compete with autotrophs, which are slower-growing organisms. This may have contributed to the increase in RMSE and MAE from the steady state values of 1.36 (both) (Table 4.4) to 6.80 and 5.74, respectively (Table 4.5).

It is clear from Figure 4.6 that the model was overestimating the effluent value of PO3-. Calibration did not seem to have a significant impact on the value, as mentioned earlier. This is also evident from the RMSE and MAE, 6.19 and 5.52 (table 4.5), respectively.

Table 4. 5 Dynamic model value for RMSE and MAE

S/N	Variable	Avg. Simulated	Avg. Observed	RMSE	MAE
1	NH_4	15.90	16.53	13.40	10.14
2	NO^{2-} & NO^{3-}	11.04	5.61	6.80	5.74
3	PO^{3-}	9.12	5.77	6.91	5.52

CHAPTER V: CONCLUSION AND RECOMMENDATION

5.1 Conclusion

This work allowed to evaluate the physico-chemical quality of the treated wastewater from the WWTP of Ain El Houtz in Tlemcen and the modeling of the biological removal process. Firstly, the Ain El Houtz wastewater treatment plant (WWTP) in Algeria was evaluated for its efficiency in removing pollutants from water. The study analyzed data from 2020 to 2022 and found that the plant performs well in removing most pollutants.

The plant has been shown to be highly effective in removing suspended solids, with a rate of 91.2%, as well as organic matter with rates of 90% and 92.2% for COD and BOD_5, respectively. These results are consistent with previous studies and meet government standards for treated wastewater intended for irrigation use. However, the plant's performance in removing ammonium nitrogen is only 60%, which is unsatisfactory. As a result, the effluent levels of ammonium nitrogen can sometimes exceed the irrigation standard. Additionally, the phosphate levels in the water are consistently higher than the recommended limit.

The biological removal process was modeled using GPS-X software and the CNPLIB library, which includes the ASM2d model for biological phosphorus removal and simultaneous nitrification-denitrification. Calibration was done using physical and operational data from the Ain El Houtz WWTP, obtained from the Office National de l'Assainissement (ONA). This data included parameters such as flow data, tank details, clarifier type, and pumped flow.

Influent characterization was carried out using the COD states model, which allowed modification of COD fractions for better WWTP model calibration. The influent advisor feature in GPS-X was employed to analyze basic wastewater metrics, including BOD5, COD, TSS, VSS, and TKN. To calibrate the model, data from the Ain El Houtz WWTP's two years (2020 and 2021) were averaged to assume a steady state. The calibration process involved adjusting COD, BOD5, TSS, NH_4, PO_3-, NO_3- & NO_2-, TKN, TP, and DO_2 parameters to match the influent and effluent data.

Sensitivity analysis was conducted on both kinetic and stoichiometric parameters. The study found that phosphorus content was insensitive to both parameters, which aligned with the results of a similar study. As a result, phosphorus content was not calibrated. However, certain kinetic and stoichiometric parameters for nitrogen content, which includes ammonium, nitrate, and nitrite ions, were found to be sensitive. Thus, six parameters were calibrated, including the reduction factor for denitrification, maximum specific growth rate for autotrophic biomass, saturation coefficient for ammonium, and oxygen saturation coefficients for heterotrophic and autotrophic biomass. The calibrated values were within acceptable ranges reported in literature.

Statistical measures such as Mean Absolute Error (MAE) and Root Mean Squared Error (RMSE) were used to evaluate the steady-state model's performance. The MAE and RMSE values for ammonium (NH_4) showed a significant difference between the simulated and observed concentrations. The simulated values for nitrite and nitrate (NO_2- & NO_3-) and phosphate (PO_3-) were higher than observed, with moderate to substantial error levels.

After calibrating the steady-state model of the WWTP, it was validated through dynamic modeling to test its predictability and forecast future events and scenarios. For this, we used data from one year in 2022, consisting of 30 individual data points. During the dynamic state model, we observed discrepancies in the NH_4 concentrations, with the model underestimating and overestimating the effluent concentration. This was reflected in the RMSE and MAE values for NH_4, which were 13.40 and 10.14, respectively. We noted similar trends in the dynamic model for nitrite and nitrate concentrations, indicating variations in denitrification rates between the model and reality, contributing to increased RMSE and MAE values from steady state to 6.80 and 5.74, respectively. The model consistently overestimated the effluent value of PO_3-, and calibration had limited impact on this parameter. This was evident in the RMSE and MAE values of 6.19 and 5.52, respectively.

5.1 Recommendation

Based on the findings of this study, the following recommendations are proposed for the Ain El Hout WWTP:

1. Enhance nitrogen removal:

To achieve complete ammonia removal, additional research is necessary to identify the underlying factors that limit the process. Some of the reasons could be the lack of adequate oxygen supply, insufficient availability of organic carbon, or the presence of specific inhibitors. After identifying these factors, it is recommended to implement advanced nitrogen removal technologies, which are specifically designed for efficient ammonia removal.

2. Address phosphate discharge:

It is essential to investigate the sources that are causing high levels of phosphate in the influent. These sources can be industrial activities, agricultural runoff, or specific household detergents. After identifying the sources, appropriate phosphate removal techniques should be explored and implemented. Some of the commonly used techniques are chemical precipitation, biological phosphate removal using enhanced biological phosphorus removal (EBPR) processes, or adsorption using specific media.

3. Refine the biological process model:

It is recommended to reevaluate the calibration of phosphorus-related parameters, considering their sensitivity to both kinetic and stoichiometric variations. In order to improve the accuracy of phosphorus predictions in the effluent, it is suggested to investigate additional phosphorus-related parameters. Finally, it is recommended to investigate the suitability of alternative models or modifications to the existing model (ASM2d) to represent the plant's specific behavior better and capture the dynamics of nitrogen and phosphate removal, such as ASM3.

4. Implement continuous monitoring and optimization:

It is important to regularly monitor the characteristics of the incoming and outgoing wastewater, paying particular attention to critical parameters such as ammonium, nitrate/nitrite, and phosphate concentrations. Based on the collected data and model simulations, it is necessary to continuously optimize operational parameters like aeration rates, organic carbon loading, hydraulic retention time, and nutrient ratios to enhance treatment efficiency.

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