MONITORING AND MODELING OF TREATED WASTEWATER RESERVOIRS DYNAMICS WITHIN THE FRAMEWORK OF AGRICULTURAL IRRIGATION

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ABSTRACT

Introduction

Quality changes that may occur in reservoirs filled exclusively or partially with treated wastewater for agricultural irrigation raise questions, especially under the climatic conditions of the Mediterranean basin. Indeed, during storage, treated wastewater is subject to several mechanisms that can influence its quality. Biological componant such as the dynamics of pathogens, the phytoplanktonic and zooplanktonic cycles, the dynamics of phosphorus and nitrogen, and the degradation of organic matter are encountered (Brissaud et al., 2000; Cirelli et al., 2008; Eme and Molle, 2013; Juanico, 1996). Physical mechanisms like thermal stratification, sedimentation, and hydraulic conditions, which can be characterized by the residence time distributions, can also influence water quality. It is important to mention also the influence of environmental conditions such as oxygenation, temperature, solar radiation, and wind, as well as the interactions between the various mechanisms mentioned above. These reservoirs are thus complex systems that can be studied by using models as a decision-support tool for predicting quality or managing these resources (Friedler et al., 2003; Mannina et al., 2008). However, these models have been poorly evaluated, and very few studies concern the storage of treated wastewater (TWW).

This study aims to characterize the effects of treated wastewater storage in open reservoirs on its quality for agricultural irrigation in the south of France, and to understand the mechanisms behind any changes observed. Our study is divided into three methodological steps: i) the monitoring of the evolution of the quality of treated wastewater stored for irrigation in the south of France; ii) the construction and validation of a dynamic model capable of representing the main mechanisms of the storage system and predicting the evolution of the quality of stored waters and iii) the study of the optimal management of TWW reservoirs through modeling. In this paper, we present the processes followed and the results obtained for the first step of the study, which is focusing on the monitoring of the evolution of TWW quality parameters in two observation reservoirs.

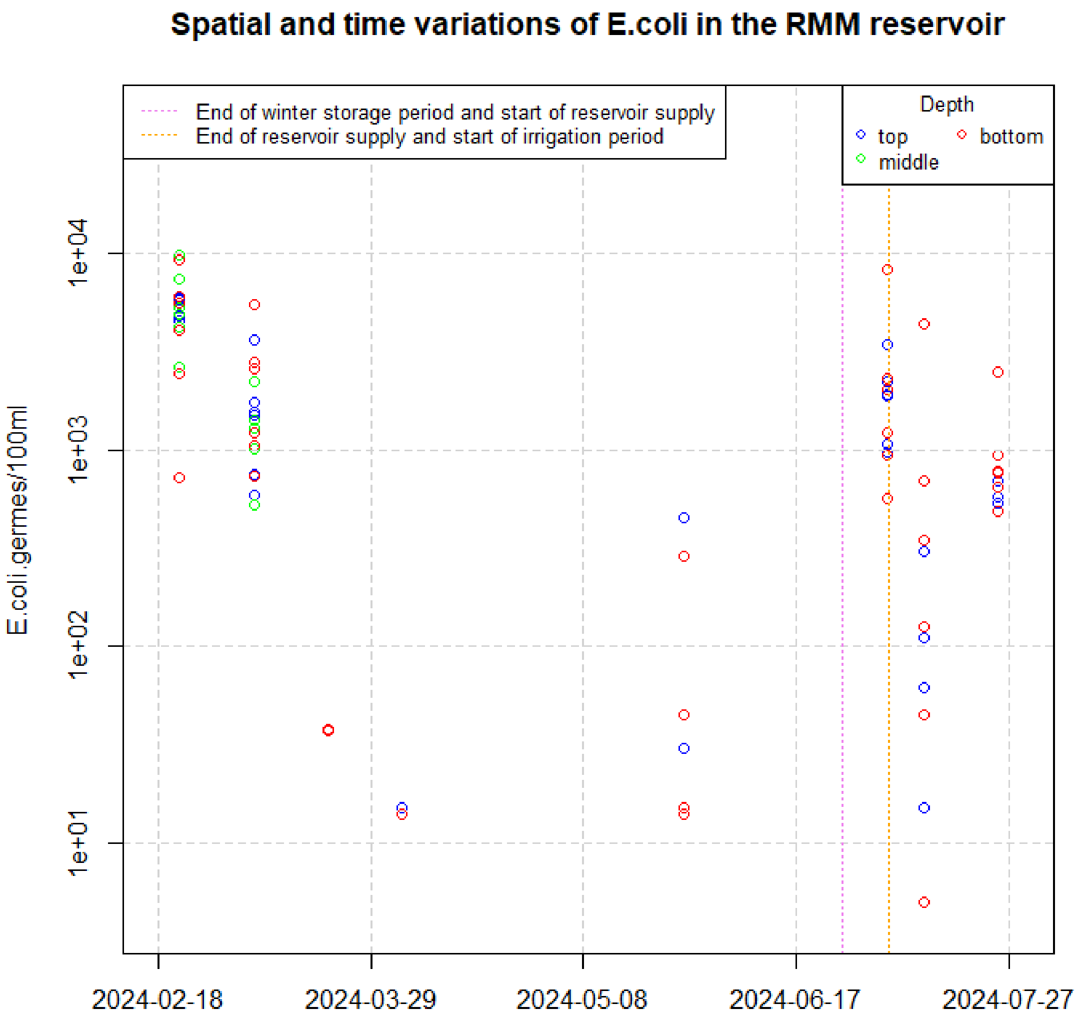
Materials and methods

The quality of TWW in this study is characterized by the set of parameters listed in the regulations regarding the reuse of TWW. The most recent decree is the one of December 18, 2023, which prescribes the expected quality level of TWW intended for agricultural use, in relation to the targeted crop types. The quality parameters in question are according to the national regulation: F-Specific RNA Bacteriophages (FSRB), *E.coli*, Spores of Sulfite-Reducing Anaerobic Bacteria (SSRAB), water turbidity, Total Suspended Solids (TSS), and Biochemical Oxygen Demand over 5 days (BOD5). These parameters have been monitored in two TWW reservoirs in the France mediterranean area, respectively associated with the wastewater treatment plants (WWTP) of Roquefort-des-Corbières and Murviel-lès-Montpellier (see appendix). In this paper, the reservoirs are designated by the "RMM" and "RRC" acronyms, which stand for Murviel-lès-Montpellier reservoir and Roquefort-des-Corbières reservoir. The following climatic, physical, and biochemical parameters were also measured: pH, dissolved oxygen, ammoniacal nitrogen (NH4-N), nitrate nitrogen (NO3-N), total phosphorus, Chemical Oxygen Demand (COD), water and air temperature, wind speed and direction, solar radiation, rainfall, water height, input and output water supply, Dissolved Oxygen (DO), and light penetration depth in the water. According to the literature, they can be useful for understanding the evolution of monitored parameters.

The choice of these reservoirs is justified by their specific geometric characteristics and the difference in water quality at the treatment plant outlet, knowing that the treatment technology used varies from one WWTP to another: activated sludge for RRC and aerated constructed wetland for RMM. The RMM and RRC reservoirs have, respectively, a maximum depths of 1.5 m and 3.7 m, with a storageapacity of around 3000 m³ of water. A system formed by a weather station and sensors connected to a data acquisition central was used to collect climatic data and some physicochemical data of the water in the basin (dissolved oxygen, water height, and temperature). Spot samples are taken for the analysis of biochemical, biological parameters, and MES. Samples are collected at 5 points on the surface of the RRC reservoir compared to 6 for RMM. They are taken at 2 or 3 positions in the water column, depending on the total water height or the spatial variations of concentrations for the concerned parameters. Samples were taken at an average frequency of once every two weeks for the pre-irrigation period and once a week for the irrigation period. We conducted 8 measurement campaigns for the RMM reservoir, including 5 during the pre-irrigation period and 3 during the irrigation period, and 5 campaigns for the RRC reservoir but only during the irrigation period.

Results and discussion

Monitoring results are given for each of the above-mentioned quality parameters, each monitoring period and each reservoir. For the RMM reservoir, monitoring of microbiological parameters for the pre-irrigation period reveals an almost total absence of FSRB for the entire measurement period, relative stability in SSRAB levels and a downward trend in *E.coli* levels over time. *E.coli* levels measured in effluent leaving the treatment plant were in the order of 10^3 to 10^4 germs/100 ml, falling to 38 germs/100 ml (limits of quantification) after around two months storage. Filling the tank during the irrigation period led to an increase in *E.coli* levels, but these fell rapidly one week after filling (see figure). The experimental set-up does not allow us to clearly identify the various factors involved in the *E.coli* degradation, but according to some authors, ultraviolet radiation and temperature play a key role in this phenomenon (Xu et al., 2002). As a result, a higher rate of degradation in summer than in winter could be suggested. But according to the figure below, the reduction of *E.coli* during irrigation period does not seem faster than during the winter period. The presence of duckweed in summer, observed in the reservoir from early July onwards, may however reduce the effectiveness of these *E.coli* destruction factors during this period (Yeh et al., 2011).



Average BOD5 values decrease over time during the pre-irrigation period. For the irrigation period, the average value of this parameter increases in the first week after the pond is filled, and then decreases two weeks later. TSS showed the same fluctuation. This could be explained by an increase of the phytoplankton biomass following an increase in the availability of nutrients (nitrogen, phosphorus) linked to pond filling. Indeed, Taffouo et al (2017) have shown that there is a good positive correlation (0.77) between phytoplankton content and BOD5. Dissolved oxygen shows the same behavior during this period. However, concentrations were higher at the surface (13 to 20 mg/L) than at depth (0 to 7.5 mg/L). This suggests an increase of photosynthetic activities, which is consistent with the hypothesis of an increase of the phytoplankton biomass as a result of nutrient inputs to the system. Although the parameters monitored may vary considerably from one point in the water column to another, the results do not reveal any spatial trends in the RMM reservoir, with the exception of dissolved oxygen. However, as this parameter was measured over a short period of the day, it probably reflects the slowdown of the photosynthetic mechanism in relation to the attenuation of Photosynthetically Active Radiation with depth (Beer Lambert's law) (Bales, 1985; Chapra, 2008). During the day, a temperature difference can be observed between the surface and the bottom of the pond, which eventually disappears during the night. There was therefore no permanent thermal stratification in the basin. This is probably due to the action of the wind, combined with the shallow depth of the reservoir (less than 1.5 m for a surface area of over 2000 m^2) (Touchart, 2002). These mixing mechanisms, which will need to be well characterized in the future, are likely to play a role in the absence of spatial trends for the various quality parameters monitored.

The RRC reservoir has undergone major hydraulic changes due to several filling-withdrawals cycles. *E.coli* and SSRAB levels appear to be influenced by this dynamic, with greater variability in the water column for these parameters during the filling phases (see figure above). In most cases, levels are higher at the surface, approaching those of effluent leaving the WWTP, i.e. 15.10^3 germs/100 ml for *E.coli*. This suggests that effluent leaving the WWTP tends to spread preferentially on the surface. The tendency towards homogenization during irrigation periods can be explained by the amplification of the mixing phenomenon linked to the currents created by pumping. In addition to this dispersion-homogenization cycle of *E.coli* levels, averages also tend to increase during the filling phase and vice versa during the irrigation phase. This could be explained by the lower water level, which favours the destruction of *E.coli* by UVs in the water column. On the other hand, as the reservoir's feed point is close to the withdrawals point, we could also suspect the existence of a short-circuit that would facilitate the aspiration of effluent from WWTPs with higher *E.coli* concentration (Eme and Molle, 2013). Average *E.coli* concentrations can go down to values of the order of 10^3germs/100 ml in the case of the RMM reservoir, while FSRB levels are below 1 germ/100 ml. TSS and BOD5 values in effluent leaving the WWTP are in most cases lower than those measured in the reservoir. BOD5 is often close to or below the limits of quantification, i.e. 3 mg O_2 /L for these effluents. This suggests that organic matter is produced by mechanisms within the reservoir. Like COD, BOD5 is almost always higher at depth than at the surface. This is probably linked to the sedimentation process or the continuous inflow of effluent from WWTPs at the surface, or to a lack of oxygenation of the water layers at depth, resulting in slower degradation of organic matter. In fact, near-bottom DO values are usually of the order of 1 mg/L, and odors were perceptible when these samples were taken, testifying to the existence of anaerobic conditions at depth. The effluents brought in are low in ammoniacal nitrogen (less than 1 mg/L), whereas up to 8 mg/L can be measured at the bottom of the reservoir. This type of nitrogen seems to derive from the ammonification of organic matter at the bottom.

Conclusion

The aim of this first stage of our study was to monitor the impact of storage on the quality of treated wastewater (TWW), more specifically on the micrological, biochemical and physical quality parameters prescribed by regulations about TWW reuse for agricultural irrigation. The storage of treated wastewater, under the study conditions, shows clearly some effectiveness in the removal of *E.coli* during the storage period, but does not appear to contribute to the removal of SSRAB. As the FSRB levels in the effluent leaving the WWTP are below the quantification limits, we were not able to monitor the effect of storage on this parameter. TSS and BOD5 tend to increase as reservoir eutrophication conditions allow. The contrasting trends in certain quality parameters suggest the probable existence of optimal situations that should be studied. A more in-depth analysis of the results of this monitoring phase and the study of these reservoirs through modeling should provide a better understanding of the mechanisms involved and their impact on the targeted quality parameters.

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Appendix

Location of the reservoirs