# Are all wetland models the same? Comparing wetland models and streamflow regulation of catchment-scale hydrological modelling tools under a changing climate

by

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# Plagiarism declaration

This dissertation reports research completed under the Department of Environmental and Geographical Sciences, University of Cape Town, between August 2019 to July 2022 for the MSc.

I, Penisoh Metho, know the definition and forms of plagiarism. I declare that all of the work in this dissertation, excluding all properly acknowledged references, is my own. All research and work presented in this dissertation has not been submitted for another degree in this or any other university.

Signed

Signed by candidate

Date: 18 July 2022

## Acknowledgements

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# Abstract

Comparing how wetlands are simulated in different hydrological modelling tools is needed to identify their suitability in different contexts. A simulated wetland will result in predictions of streamflow regulation, e.g., storing flood water and reducing high flows and releasing water in drier periods, which may or may not be realistic for a given area. Evaluating wetland models is critical for navigating the different types of physical wetlands with variable influences on streamflow, and the different simulated wetlands conceived in the plethora of modelling tools (i.e. software) available for use. A recent study found that sometimes wetlands are excluded from hydrological models used to inform water resource decisions. When wetlands are included in a hydrological model, few studies identify process similarities between the actual and modelled wetland or the realism of the modelled impacts of the wetland on streamflow before applying the model's output to water resource decisions. This research aims to identify and evaluate wetland characteristics, processes and impacts on catchment streamflow in different modelling tools and models (i.e. setups in a tool). Evaluating wetland models supports wetland-inclusive modelling and ensures that a wetland model is hydrologically sound and suitable. An unchannelled valley-bottom wetland located in the upper Kromme catchment, Eastern Cape, South Africa, was used. Wetland models were compared as independent units conceptually and as functional units within the catchment by modelling. First, using qualitative analysis, a conceptual assessment of wetland model structures in ACRU, WRSM-Pitman, MIKE SHE coupled with Hydro River and SWAT were considered in the context of the case study wetland. Second, using quantitative analysis, model outputs from wetland models in ACRU and WRSM-Pitman were assessed for model performance, behaviour and streamflow regulation during droughts and floods. The predicted impact of the wetland on catchment hydrology was determined from scenarios with and without a wetland and modelled wetland storage fluxes over the whole simulation period, four severe floods and three drought periods.

The results from the qualitative and quantitative comparisons suggest that similarities between the physical and simulated wetland improves the likelihood of model suitability, good model performance and streamflow regulation predictions. Additionally, models setup for the same wetland with the same input data simulated potentially acceptable but different streamflow totals: for an observed total of 9.13 $Mm^3$; WRSM-Pitman's comprehensive wetland simulated 10.64 $Mm^3$; and from ACRU's riparian zone and wetland HRU's simulated 11.31 $Mm^3$ and 8.89 $Mm^3$, respectively. Modelled actual evapotranspiration was underestimated by the riparian zone wetland (946.08 mm), overestimated in the comprehensive wetland model (2 054.80 mm) and moderately similar in the wetland HRU when compared with remotely-sensed data (1 520.30 mm). During extreme events, all models simulated flood attenuation while drought responses were variable (two wetland models predicted streamflow attenuation). By implication, the results suggest that good model performance does not guarantee the simulation of expected streamflow regulation roles recorded in literature. Furthermore, variable water yields and wetland impacts from the models demonstrated the possibility for different modelling efforts to result in different water supply, use and conservation measures. The study highlights the importance of contextualising model output for catchments with wetlands before applying the simulations to impact assessments or future climate scenarios.

***Keywords:*** wetlands; streamflow regulation; intercomparison; hydrological modelling; floods; droughts; climate extremes

# i. List of abbreviations and acronyms

| Abbreviation | Description |
|---|---|
| **ACRU** | Agricultural Research Catchment Unit |
| **AET** | actual evapotranspiration |
| **AWS** | Automatic weather station |
| **CV** | Coefficient of variation |
| **CW wetland** | comprehensive wetland module representing a wetland in WRSM-Pitman |
| **CW model** | WRSM-Pitman model with the comprehensive wetland (catchment & wetland) |
| **DEM** | Digital elevation model |
| **ET** | evapotranspiration |
| **FDC** | flow duration curve |
| **GRA** | Groundwater Resource Assessment |
| **HRU** | hydrological response unit |
| **KGE** | model performance metric, Kling-Gupta efficiency |
| **MAP** | Mean annual precipitation |
| **MAR** | Mean annual runoff |
| **MS-MHR** | MIKE SHE coupled with MIKE Hydro River |
| **NSE** | model performance metric, Nash-Sutcliffe efficiency |
| **NSE (log Q)** | model performance metric, Nash-Sutcliffe efficiency using log transformed input that reduces biases on high flows |
| **PET** | Potential evapotranspiration |
| **RZ wetland** | riparian zone HRU representing a wetland |
| **RZ model** | ACRU model with the specialised riparian zone HRU (catchment & wetland) |
| **SD** | Standard deviation |
| **SEBAL** | Surface Energy Balance Algorithm for Land |
| **SPEI** | standardised precipitation evapotranspiration index |
| **SRTM** | Shuttle Radar Topography Mission |
| **SWAT** | Soil and Water Assessment Tool |
| **WL wetland** | wetland HRU representing a wetland in ACRU |
| **WL model** | ACRU model with the specialised wetland HRU (catchment & wetland) |
| **WRSM-Pitman** | Water Resources Simulation Model-Pitman |

# ii. Glossary of terms

| Term | Description |
|---|---|
| **HRU** | conceptual aggregation of areas with the similar properties and hydrological responses |
| **Modelling tool** | the software from which hydrological models can be constructed and run;<br>a modelling tool often refers to a hydrological model in other literary sources, but it is ambiguous for indicating whether it is referring to the software or the methods and processes expressing the catchment hydrology with or without modeller choices (e.g. configuration- or parameter-related) |
| **Model** | a concept or perception of how things work or fit together; the configuration of the catchment or wetland in a specific modelling tool which is a combination of the model structure and modeller choices |
| **Model structure** | (1) the context in which the model calculates the storage and flows of water in the modelled domain;<br>where context refers to the suite of spatiotemporal descriptions, catchment processes, characteristics, and algorithms used to describe the catchment hydrology in a modelling tool or a specific setup of a modelling tool;<br>(2) the concepts, algorithms and parameters which describe the catchment hydrology and processes in a model, excluding modeller choices |
| **Strategic water source area** | upstream areas which provide a significant amount of water to downstream economies and economic centres;<br>these areas supply a disproportionate amount of mean annual runoff to another area of interest (see Nel et al., 2013) |
| **Wetland model** | interchangeable with simulated wetland, refers to the wetland unit from a hydrological modelling tool |
| **Wetland representation** | the extent to which a modelling tool or setup covers the qualitative and quantitative characteristics and processes of a physical wetland; wetland representation is achieved if the characteristics, processes, and role of the physical wetland in the catchment are included in the modelling tool or model |

# Chapter 1: Introduction

## 1.1. Introduction

There are many different catchment hydrological modelling tools available, many of which include subroutines to represent wetlands. Different software tools use different approaches to modelling wetland processes, which may be more or less appropriate for a specific wetland in a specific catchment. There are many types of wetlands, such as floodplains, valley bottoms, pans, and seeps. Wetlands differ in geomorphology, surface and subsurface flows and storages, and resulting impacts on streamflow. Wetland models in software tools often represent a particular, generic conceptualisation of a wetland, translated into algorithms governing inflow, storage, and outflow. These will differ in their capability to represent the dominant processes of different wetlands. As such, consolidated information on how various modelling tools represent wetlands can assist in model selection and set-up choices.

There is growing recognition of the need to consider the streamflow regulation impacts of wetlands in hydrological modelling. Explicitly including wetland flows and storages can improve catchment model accuracy in general, and wetland models can assist us in predicting the impacts of losing or restoring wetlands, thereby influencing catchment management decisions. However, because different models represent wetland processes in different ways, the likely realism and suitability of models for different wetland use cases needs to be evaluated.

This research aims to compare wetland model characteristics, process representation, and streamflow regulation predictions across several commonly used modelling tools in South Africa, in the context of a particular wetland case study: an unchannelled valley bottom wetland. This chapter provides an introduction to the dissertation by first, noting the background information and context of wetlands modelling and wetlands in South Africa; second, stating the research aims, objectives and questions; and thirdly, concluding the chapter with the significance and limitations of the study.

## 1.2. Background information and foundational concepts

### 1.2.1. The value of wetlands in policy, practice and hydrology

Wetlands in South Africa are severely degraded yet essential to the integrity and functioning of the environment and society. In early to late 1990's, wetlands were regarded as wastelands and unproductive land fit for transformation into agricultural land or other uses with immediate economic gains (Matthews, 1993). This ideology resulted in the destruction of more than 50 % of South Africa's wetlands (Cowan, 1995) and more than 65 % of the remaining wetlands classified as threatened ecosystems (Nel and Driver, 2012). At the turn of the century, the Millennium Ecosystem Assessment helped global thought transition from the narrative of "wetlands as wastelands" and replaced it with a more favourable, empirically proven narrative where wetlands are both economically and environmentally valuable (MEA, 2005). This transition coincided with the introduction of the integrated water resources management and increasing appreciation of the benefits from healthy environments (NWA, 1997; NEMA, 1998). As a result, South Africa leans towards unsubscribing from the narrative of wetlands as wastelands. With wetlands identified as providers of ecological services, a recent study estimated South African ecosystem services as contributing U$ 610 billion which is 1.5 times greater than the 2014 GDP (Anderson *et al*., 2017).

South African legislation considers wetlands as high value natural resources. At the inception of an international and intergovernmental treaty, the Ramsar Convention, South Africa voluntarily joined as a participating signatory (Ramsar, 1971). The treaty aims to promote the conservation and wise use of wetlands through local, regional and international cooperation (Ramsar Convention Secretariat, 2018). As a signatory of the treaty, several laws regarding the conservation and equitable management of wetlands have been introduced. For example, influences from this enrolment are evident in the National Water Act (NWA, 1998), the National Environmental Management Act (NEMA, 1998), and National Environmental Management Biodiversity Act (NEMBA, 2004). In the case of water resources and wetlands, the policies attempt to stay relevant and dynamic with phased updates and revised goals in the National Water Resources Strategies (DWA, 2013), technical papers (DWA, 2014) and management guidelines (DWS, 2016; MacFarlane *et al*., 2014; MacFarlane and Bredin, 2017). In terms of wetland extent regionally appreciated, there are currently twenty-eight wetlands of international importance and counting in South Africa (Ramsar Sites Information Service, 2021) among other nationally defined wetlands (van Deventer *et al*., 2020).

In addition to policy, the value of wetlands is reflected in practice. The importance of wetlands is evident in two rehabilitation initiatives instituted by the national government and implemented at grassroots levels: namely, these are the Working for Water programme and the Working for Wetlands. The Working for Water program focuses on clearing invasive alien plants while the Working for Wetlands program aims to rehabilitate wetlands (van Wilgen *et al*., 2012; Working for Wetlands, 2005 and updates). In addition to this, there are several examples of private, public and civil partnership and efforts to address wetland monitoring, conservation and the ecological services provided by wetlands in rural and urban areas (Aurecon, 2019; Kotze and Ellery, 2009; Mander *et al*., 2017; Nemutamvuni *et al*., 2020; Sieben *et al*., 2017; 2021; Turpie *et al*., 2017; Belle *et al*., 2018). Furthermore, the conservation and management of wetlands takes into consideration that there are different types of wetlands which results in every wetland having different vulnerabilities, management requirements and protection actions (DWA, 2014). In the instances of land use impacts, creating buffer zones around wetlands and riparian habitats attempts to be wetland-specific instead of generalised (NWA, 1998; MacFarlane and Bredin, 2017). A wetland-specific approach is also used for setting the management objectives for wetlands and quality to maintain (Bredin *et al*., 2019) and managing disaster risks (Belle *et al*., 2018). Fortunately, these efforts provide evidence for actions towards preserving wetlands and their ability to provide ecological services.

From a hydrological perspective, the water provisioning and regulation are the most important ecological services from wetlands. Categorically, wetlands have services which may provide a benefit, regulate a natural process, support natural processes or contribute to human wellbeing (MEA, 2005). Respectively, these are provisioning, regulating, supporting and cultural ecological services. Provisioning services in favour of the local hydrology is the water stored in wetlands. Wetland water storage contributes to the local water supply for communities, agriculture and livestock. Sometimes, wetlands can also be a source of water losses with high evapotranspiration rates from the vegetation within the wetland. However, in most cases, wetlands are an additional water supply and water loss simultaneously. In terms of regulation, wetlands regulate streamflow and water quality. Generally, most wetlands have vegetation, soil and land surface properties which promote water retention and flow velocity reduction. These properties enable the wetland to intercept flood waters and notably increase low flows with the release of water from the wetland during low rainfall or river flow (Mitsch *et al*., 2009; 2015; Acreman *et al*., 2003; Kadykalo *et al*., 2016). This is referred to as streamflow regulation and occurs in contrast to the presiding climate conditions (for example, a wetland may reduce the river flows associated with floods and increase low flows associated with droughts). Considering South Africa's semi-arid climate, limited water resources and high climate variability (Schulze, 2012), wetlands contribute to the resilience of water resources against the negative impacts of climate on water availability. During high flows, wetlands reduce flood pulses and the associated damage, and store water. During low flows, wetlands may increase water availability. Consequentially, wetlands are referred to as ecological infrastructure which support water security (Bonthuys, 2018).

### 1.2.2. Navigating wetland modelling

It is common for modelling studies to use wetland models as they are in the modeller's preferred catchment modelling software (interchangeably and hereafter referred to as the modelling tool) without evaluation of wetland model realism. In addition to the modeller's familiarity with the modelling software, modelling tools are often selected based on the model's skill for the impact assessment (i.e. land use change, climate change) or level of detail in the tool (i.e. scale of the catchment or temporal resolution of the output). From earlier to later modelling tools, recognition of the importance of wetlands is evident in the creation and development of wetland routines. Initially, this involved the addition of wetland routines to existing catchment models. Three locally-developed and commonly used tools have added or modified wetland routines. These tools include Pitman-SPATSIM (Hughes *et al*., 2013), WRSM-Pitman (Pitman *et al*., 2000; Bailey and Pitman, 2016) and ACRU (Schulze, 1995 and updates, Gray 2011; Thornton-Dibb *et al*., 2010). More recently, wetland modelling studies are focusing on improving and testing the existing wetland models within one modelling tool (Evenson *et al*., 2016; Rahman *et al*., 2016; Gray, 2011; Pitman and Bailey; 2015; Qi *et al*., 2019; Hughes *et al*., 2013). Less effort has been put into comparing wetland models from different tools or relative to real wetlands. An exception has been the work of Maherry *et al*. (2017), deriving twenty-one wetland concepts to guide the setup of wetland types in any modelling tool and summarising wetland models from seven modelling tools. There are several examples of applying wetland models, as available in modelling tools, to wetland loss, degradation and climate change impact studies. For example, modelling has been used to see the how the catchment runoff changes in response to wetland degradation, rehabilitation, and other land use changes in a catchment (Rebelo, 2012; Rebelo *et al*., 2015). There are also examples of case studies assessing the influence of wetlands on water availability in historical and future climates (Gray, 2011; Fossey and Rousseau, 2016). If wetland representation is to be accurate and maximised, using wetland models without reality checks is problematic and may lead to misrepresented wetlands. By incorporating reality checks for wetland models, a modeller is able to assess and evaluate wetland representation, improve a wetland model where necessary and choose the most appropriate model for a specific wetland type with unique environmental settings, properties and influences on streamflow.

Difficulties related to standardising wetland model comparisons are the main barriers to comparing wetland models. Ideally, the criteria for comparing wetland models would be applicable to all models involved. However, wetland models have different contexts and spatiotemporal scales.

Firstly, wetland models within a tool tend to be developed for a specific wetland type. Some routines are developed for riparian wetlands and others geographically isolated wetlands. As an example of the diversity of wetland models, within one tool separate routines are available for wetlands which are similar to lakes, paddies, or reservoirs.

Secondly, across modelling tools, wetland models are differentiated by how they conceptualise the wetland's water balance, their temporal scale and their connection with the surrounding catchment and river. Cumulatively, this leads to wetland models with different processes and variables. Temporally, modelling tools may simulate the catchment at monthly to sub-daily scales. Spatially, the catchment or wetland may be simulated at sub-catchment to regional scales.

Thirdly, modelling tools, and by inheritance the wetland models, may also differ in their level of complexity ranging from conceptual models to models replicating physical laws governing water flows and movement.

As a result, wetland models have different contexts which makes it challenging to compare a detailed, sub-daily to daily wetland model with simpler models simulating the wetland water balance as coarser time scales. In addition to this, the model runoff and water balance components may be different between tools. This complicates how to compare output from different tools. Altogether, these differences frustrate efforts to compare wetland models, identify equally applicable metrics and increases the time needed to do a comparison.

### 1.2.3. The case for explicit inclusion of wetlands in catchment modelling

In addition to navigating wetland model differences, some studies in water resource management have been reported to exclude explicit representation of wetlands in their models for catchments with wetlands. According to Maherry et al. (2017), practitioners often do not include any specific consideration of wetlands in hydrological models used to inform water resource management. The study highlighted that not explicitly modelling wetlands in hydrological models gives modelling results that are not representative of actual catchment conditions and have incorrect hydrological reasoning deriving the output. The risks associated with this approach could be amplified and are concerning when considering the application of hydrological models to planning, managing and informing water resource decisions; and the potential to over allocate water resources which are already limited.

The availability of resources and expertise appear to be the primary barriers to wetland-inclusive modelling. On one hand, modelling and model development requires a large amount of data. At the moment, South Africa's streamflow records are sparse with decommissioned and damaged stations declining the extent of the monitoring network (Okello et al., 2015; Pitman, 2011; Wessels and Rosseboom, 2009). Therefore, the available streamflow records are sometimes insufficient to support most long-term modelling activities at fine scales or with recent data. In the case of data records for wetlands, data scarcity is even more severe for wetland inflows, outflows and storage which are necessary for developing and testing whether wetland models are accurate. Wetland monitoring is usually available and concentrated in a few wetland sites where researchers are conducting studies. These data limitations restrict wetland-inclusive modelling since there is minimal data to validate the model output and confirm if the model setup is acceptable. On the other hand, expertise and familiarity with a modelling tool can allow the exclusion of wetlands from modelling tools. According to Maherry et al. (2017), when wetlands were excluded, modellers reported adjusting other parameters to incorporate the effect of wetlands on streamflow.

### 1.2.4. Wetland representation in the model selection process

Considering the use of hydrological models in impact studies and an increasingly variable climate (Schulze et al., 2012), wetland inclusive modelling and accurate process representations are essential for planning and managing water resources. The selection of a model and modelling tool for project is typically a compromise across multiple factors, such as the input data requirements vs data availability or the capability of the model to output a variable of interest. The realism of the model's representation of wetlands specifically is not often actively considered in this process. Realistically, when selecting a model to inform the management of resources, only a few factors can be prioritised and optimised in the model selection process. According to Kundewicz *et al.* (2019), several technical factors guide the model selection process (Figure 1). Collectively, the factors determine whether the modelling tool can represent a catchment's properties, the scale and purpose of the study, and the ease of using the modelling tool. Ideally, all factors would be prioritised. However, not all factors can be optimised due to time and resource restraints. Modelling a wetland with accurate process representation could be categorised with technical factors relating to the catchment properties and purpose the study. A recent survey found that modelling tools are generally chosen for their ease of use and familiarity with the tool (Glenday *et al*., 2021). In addition to this, once a modelling tool has been selected, the model configuration and calibration usually favours replicating a section of the hydrograph (for example, high or low flows) (Lane *et al*., 2019). This part of the modelling process is also open to subjective differences from a modeller’s choices. Together, the technical factors as well as current approaches in model selection and setup suggest that wetland representation is not a priority. This also means that the potential differences between modelling tools, models and physical wetlands are currently not accounted for.

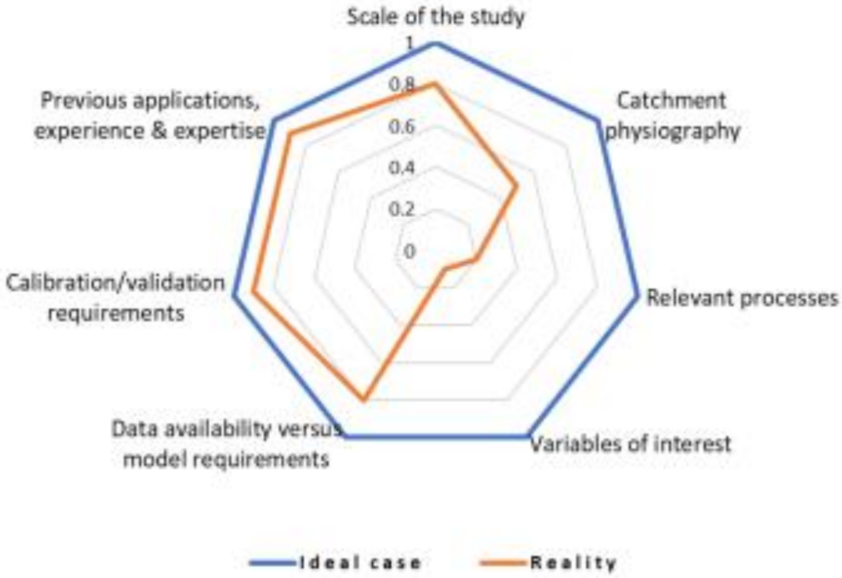



*Figure 1. Technical factors to optimise in the process of selecting a modelling tool (demonstrating priorities in ideal and actual approaches)*

### 1.2.5. Modelling streamflow regulation

Streamflow regulation is an ecological service from wetlands which is closely linked to the relationship between climate and available water resources. Streamflow regulation includes (often associated with floods) or supplement low streamflow (associated with drier seasons and, in extreme cases, droughts). Following the MEA highlighting several ecological services of wetlands, streamflow regulation recently received global publicity again. Every year the Ramsar Convention and contracting parties select a theme to be celebrated on World Wetlands Day and prioritised in research. Within the duration of this thesis, the annual theme and focus on wetlands addressed the relationship between wetlands and climate change in 2019 and the relationship between wetlands and water resources was highlighted in 2021 (UNESCO, 2019; 2021). Continued interest in streamflow regulation over the years demonstrates the relevance and importance of streamflow regulation, understanding the process and quantifying the service.

Modelling streamflow regulation requires navigating its variability. Streamflow regulation varies with wetland type. Different types of wetlands will have different net impacts on streamflow due to their differing inflows, storages, and outflows. The classification of wetlands as hydrogeomorphic units is hydrologically relevant for describing wetlands and their streamflow regulation by distinguishing wetlands based on the landscape positioning, hydrodynamics and hydrological function, there are seven types of wetlands (Ollis et al., 2016) and each wetland type affects streamflow differently depending on the season and variable extents (Mbona et al., 2016).

Similarly, wetland type and the impact on streamflow can also be related to the wetland's location in the landscape. Initially, riparian wetlands were thought to have a greater influence on streamflow regulation based on the larger upstream areas draining into the wetland and the riparian wetland's consistent connection and close proximity to the main channel, but this has since been disproved with studies showing the wetlands further away from the main channel can have considerable to equal impacts on the streamflow regulation (Yeo et al., 2019a; Lee *et al*., 2018a; Blanchette *et al*., 2022).

Furthermore, streamflow regulation may also vary with the scale of wetlands in consideration. One school-of-thought considers wetlands as individual wetlands or a complex made of several, connected wetlands (Brannen et al., 2015; Haigh et al., 2002; Mukherjee and Pal, 2021; Haque et al., 2021). Another perspective considers the size of the wetland as a determining factor of the potential streamflow regulation (Acreman et al., 2007). For example, small or local wetlands will have minimal impacts on streamflow while larger wetlands in larger catchments have a greater impact on streamflow. In the WET-EcoServices tool, a similar view on wetland size affecting the regulation is present with the catchment-wetland size ratios which are used to determine the importance of the interventions to enhance the wetlands services (Kotze et al., 2008 and updates). According to some studies, in some cases, the landscape aspect overrides whether streamflow regulation is considered from individual wetlands or wetland complexes (Helmschrot, 2006;2007; Hilbich et al., 2007; Dahlke et al., 2005). With different scale considerations and potential implications on streamflow regulation, the current perspectives suggest that the effect of scale needs to be assessed on a case-by-case basis.

Either way, for any wetland type, it is possible for a given wetland to either perform one or both of the streamflow regulation roles. Additionally, the preceding wetland storage may also affect the streamflow regulation. In such cases, there is evidence for wetlands having no impact on streamflow or regulatory influences contrary to the roles listed in literature (Riddell *et al*., 2013; Acreman and Holden, 2013; Salimi et al., 2021). Furthermore, another study showed that the streamflow regulation may vary with how pristine or impact a wetland is. Rebelo et al. (2019a) found that streamflow regulation from the Kromme unchannelled valley-bottom wetlands was significantly reduced in impacted wetlands or where wetland area was lost compared to the upstream wetlands which were relatively intact and pristine wetlands.

Within the wetland model, streamflow regulation needs to be captured in the processes describing how the wetland fills with water or releases water (i.e. spill-and fill dynamics), the connectivity of the wetland with other wetlands or water bodies (i.e. surface water connectivity (Liebowitz et al., 2016; 2018) and the relationship between surface water and groundwater (Calhoun et al., 2017; Ameli and Creed, 2017). This requires detailed information on the wetland storage and properties regulating the inflows and outflows.

In terms of defining the metrics for assessing streamflow regulation, there are many metrics which can be used. The change in runoff between scenarios with and without a wetland and flow-specific analyses are commonly used (Wu et al., 2019; Fossey and Rosseau, 2016; Rebelo *et al*., 2015; Bai et al., 2021). This is in contrast to earlier wetland modelling which focused on evaluating the water balance of intensively monitored wetlands (Acreman et al., 2003). Less commonly used metrics include time or flow specific assessments related to return periods (Wolski *et al*., 2006; Fossey and Rosseau, 2016; Mandlazi, 2017) or long-term seasonality in the hydrological year and streamflow simulations from dry and wet years have been used to detect the wetlands impact on water availability (Euser *et al*., 2013; Wu *et al*., 2009). Currently, according to a meta-analysis of studies investigating streamflow regulation services, Kadykalo and Findlay (2016) found that impact metrics are specific to the research question of a study and results in different metrics being used in different studies. Across different studies, different metrics of streamflow regulation is a barrier to comparing streamflow regulation reported from different studies.

## 1.3. Research problem

Streamflow regulation by wetlands affects water availability. In the hydrological models used to determine water availability and the impact of different environmental changes on streamflow, wetland models vary. At the same time, streamflow regulation varies temporally, spatially and according to the wetland type.

However, despite these differences, there are no comparisons of wetland models and their predicted responses during extreme events for South African catchments. Additionally, wetland models are yet to be assessed relative to a case study wetland to determine whether setting up wetland in different tools and models produces the same streamflow volumes and streamflow regulation. Furthermore, there is no standardised way to gauge whether the wetland model is appropriate for a given wetland type prior to modelling.

Therefore, there is a possibility for different models and outputs to lead to different water supply estimates and management interventions. In addition to this, unsuitable wetland models may be applied to modelling a specific wetland type. Lastly, current studies are not typically assessing the model's behaviour, or predicted streamflow regulation, during historical extreme conditions prior to applying the models to scenarios of change.

## 1.4. Research aims, questions and objectives

Considering that there a few multi-model, comparative studies focusing on hydrological modelling of wetlands, the impacts on the streamflow and accurate process representations, this study aims to identify and evaluate wetland characteristics, processes and impacts on catchment streamflow (i.e. hydrological function) in different modelling tools and models.

The main objective of the project is to identify most suitable model setup for a case study wetland typed as an unchannelled valley-bottom wetland. This will be achieved in the following sub-objectives:

***Research objective 1***:

To investigate and compare hydrological characteristics and processes defining wetlands across a set of commonly used hydrological modelling tools, in particular reference to representing an unchanneled valley bottom wetland (viz. comparing wetland models conceptually without performing quantitative predictive modelling)

***Research objective 2***:

To determine and compare the modelled impacts of an unchanneled valley-bottom wetland on streamflow over the whole simulation period and during extreme events (e.g. droughts and floods) *(viz. quantitative comparison of wetland models)*

***Research objective 3***:

To compare the model suitability derived from model concept comparisons in objective 1 with the model suitability derived from modelling the case study wetland (with particular reference to the model performance and streamflow regulation) in objective 2

Therefore, the research questions of this study are as follows:

1. How do different modelling tools conceptualise an unchannelled valley-bottom wetland?
2. What impact do the simulated wetlands have on modelled catchment streamflow during the whole simulation period, floods and droughts?
3. How does wetland model suitability, as assessed based on a conceptual review of model structure, compare to quantitative assessments of models' hydrological flux predictions?

## 1.5. Importance of the research

This study will contribute to the body of knowledge on model intercomparison projects by demonstrating a method to compare wetland models relative to a real wetland without modelling; and identifying whether different models predict the same streamflow volumes and streamflow regulation from modelled output.

This will help address the current shortage of research in this wetland model comparisons and wetland modelling applied to impact assessments. The outcomes of this research will provide value to practitioners and modellers in industry, research and model development who may need the information or method to firstly, inform their model selection process in a way the prioritises accurate wetland representation for wetlands within the catchment; secondly, support strategic monitoring; and thirdly, apply models which are credible to management decisions and impact assessment modelling.

## 1.6. Research scope and limitations

Concerning the scope of the dissertation, these objectives and research questions were obtained, and will be applied to wetland models, from a selection of modelling tools developed internationally and in Southern African. For high impact and relevance, modelling was completed on tools which indicated high suitability for the case study wetland in the first objective and were developed and commonly used in South Africa. A qualitative comparison will compare the wetland models inter of wetland characteristics and processes. Implementing the wetland model into the catchment context, modelling will quantitatively compare wetland models in terms of the model performance and streamflow regulation. All comparisons will be relative to the case study wetland as opposed to other modelling tools. Therefore, the comparisons and results are intended to reflect model capabilities for a specific wetland instead of defining any modelling tool as better or worse than others.

Some limitations which may arise from this approach is the limited scope in terms of modelling tools, models and one type of wetland (therefore, climate and landscape conditions) being investigated. For example, out of many modelling tools available for use, only four are considered in this study for one wetland type in one location and environment.

In addition to this, extreme climatic events will be constrained to historical events and the data quality of the associated records. Model behaviour which is consistent with the literature expectations of streamflow regulation for an unchannelled valley-bottom wetland and similar to the observed data does not guarantee that the models will continue to be accurate in future climates. In addition to this, extreme events are not stationary such that the selected floods events and drought periods may be equalled or exceeded in the future. Streamflow regulation and wetland model responses are not considered for future extreme evens.

## 1.7. Outline of the dissertation

In Chapter 1, the context of the research was introduced. The research rationale, aims and objectives were identified. Additionally, the scope and limitations of the study were acknowledged.

Following this introductory chapter, the dissertation is structured into four sections.

In Chapter 2, the literature defining physical and simulated wetlands (interchangeable with the term wetland models) are presented to derive the boundaries for wetland representation.

In Chapter 3, a qualitative assessment of wetland representation is conducted. The chapter offers a review of wetland model structures and compares their setups for a case study wetland.

In Chapter 4, the case study wetland is modelled providing a quantitative assessment of wetland representation.

In Chapter 5, the conclusions of the study are summarised.

# Chapter 2: Literature review

## 2.1. Introduction

The research topic of this dissertation is comparing different modelling tools and models (i.e. different software and configurations). As a starting point to the dissertation, this literature review introduces the key definitions and themes for wetlands and wetland modelling. In doing so, the literature review serves the following purposes:

- Provides standard definitions for physical and simulated wetlands
- Highlights the main features of simulated wetlands which can be compared between models and relative to a physical wetland
- Illustrates the state and development of wetland modelling
- Identifies how the predicted impact of wetlands on catchment streamflow is accounted for in hydrological modelling studies
- Identifying how wetlands are represented in hydrological models

In terms of scope, the literature review considers wetland modelling in hydrological models that can be applied to catchment-scale estimates of water availability.

The following literature review is structured into three sections. First, the review begins with an overview of physical wetlands: clarifying the definition used in this study and the hydrological processes of wetlands underpinning their importance to water resources. Secondly, the review explores simulated wetlands and their applications to date. A definition for simulated wetlands (interchangeable with wetland models) is presented together with the essential and differentiating features of wetland models. Lastly, the literature review concludes with the key findings and research gaps in wetland modelling, and the implications for comparing wetland models.

## 2.2. Physical wetlands

### 2.2.1. Definition and variability of wetland types

There are several wetland definitions which sometimes causes confusion regarding what is (or isn't a wetland). Wetlands definitions differ according to the source. Considering the literature referred to, wetlands can be defined from classification systems based on objective and empirical features, typologies based on conceptual understanding of the wetland and expert opinion, regionally, or by organisations (Gerbeaux *et al*., 2018). Concerning where the term is defined, globally-defined, interdisciplinary wetland definitions are inclusive of many water bodies and sometimes include land features such as caves (Ramsar, 1971); and may differ by region or institution (see EPA, 2022; UNESCO, 2021; IPBES, 2019). Some are internationally acclaimed and adhered to definitions are presented in the broad classification system from Ramsar's Convention (Ramsar, 1971) and the classification of wetlands and deep-water habitats of the United States (Cowardin *et al*., 1995). Other classification systems have developed regionally to account for the local environmental settings, vegetation, landforms, water regimes, and policy and decision-making contexts with examples from India (Das and Pal, 2018), Brazil (Junk *et al*., 2018), South Africa (Ollis *et al*., 2013), United States (Brinson, 1996; Tiner, 2018) and Australia (Semeniuk and Semeniuk, 2011). Across the different sources and definitions, there is consensus that wetlands have some level of permanent saturation, hydric soils and adapted vegetation, and unique water regimes.

The definition used in this research refers to the wetland definition in South African legislation and commonly cited in literature. Environmental studies and water-related polices in South Africa rely on the national legislation in to define wetlands. The National Water Act (Act No. 36 of 1998) (NWA, 1998) defines wetlands as:

"land which is transitional between terrestrial and aquatic systems, where the water table is usually at or near the surface, or land which is periodically covered with shallow water, and which land in normal circumstances supports or would support vegetation typically adapted to life in saturated soils".

Locally, this definition was used in the classification system for wetlands and other aquatic ecosystems in South Africa (Ollis *et al*., 2013) and applicable to wetland in drylands (Tooth, 2018). Furthermore, this definition is similar to international literature which defines wetlands on the similar basis of long-term saturation and anaerobic-adapted (or hydrophilic) plants and biological activities (Mitsch *et al*., 2009; 2015). Using the same definition of wetlands as the one provided in national institutional integrates the term into existing research and perspectives in policies.

Within the reference classification system (Ollis *et al*., 2013), additional wetland differentiating features useful for hydrological studies are present. The classification of wetlands as hydrogeomorphic units (HGM) is widely used and hydrologically relevant. In this framework, the HGM classification level classifies wetlands by their landform (e.g. wetland shape and topographical location), hydrological characteristics (e.g. the movement of water in and out of the wetland) and the hydrodynamics of the wetland (e.g. the predominant flows through the wetland) (Ollis *et al*., 2013). The HGM level of classification has seven types of wetlands including floodplains, channelled and unchannelled valley-bottoms, seeps, depressions and pans which are distinguished from rivers, the seventh type of wetland in the classification. Each wetland type has a unique topographical setting and hydrological processes. This demonstrates that there are many types of physical wetlands. In practice, wetlands typed as HGM units has been applied to several modelling studies (Rebelo *et al*., 2015, Maherry *et al*., 2017; Tanner *et al*., 2019) assessments of ecosystem services (Rebelo *et al*., 2019a), wetland mapping (van Deventer *et al*., 2018; Rivers-Moore *et al*., 2020; Le Roux, 2020) and field assessment procedures (Kotze *et al*., 2018; Kotze *et al*., 2019). According to the reference classification and extensive application of HGM classified wetlands, not only are HGM wetlands hydrologically relevant (i.e. describing the context and movement of water in, through and out of a wetland), but it covers the range of possible physical wetlands. This comprehensive description and coverage of physical wetlands as HGM units promotes its usability for hydrological modelling and as a basis of differentiation between wetlands.

### 2.2.2. Influence on water availability

Different physical wetlands have different influences on streamflow. Streamflow regulation refers to the ability of wetlands to attenuate high streamflow (often associated with floods) or supplement low streamflow (associated with drier seasons and, in extreme cases, droughts) (Acreman *et al*., 2003; Kadykalo *et al*., 2016). This is an ecosystem service from wetlands which increases the resilience of water resources to climate-induced threats to water availability, allows wetlands to be a source of water and reduces the hazards accompanying floods. However, not all wetlands attenuate and supplement streamflow. A recent expansion on the classification of wetland HGMs presented how the extent, timing and impact on streamflow varies by wetland type (Mbona, 2016; Table 1). As a result, it is possible for a given wetland to either perform one or both of the streamflow regulation roles. In some cases, there is also evidence for wetlands having no impact on streamflow or influences contrary to the roles listed in literature (Riddell *et al*., 2013; Acreman and Holden, 2013; Salimi et al., 2021). Similar findings of streamflow impacts from wetlands differing by wetland type, namely riparian or geographically isolated (i.e. wetland type based on the wetland's connection and location relative to the river) have been reported in international literature (Lee *et al*., 2018a; Yeo *et al*., 2019a). Generally, riparian wetlands are thought to have a greater impact than geographically isolated wetlands, but these studies advocate for the potential of geographically isolated wetlands to regulate streamflow comparably with riparian wetlands. In terms of determining whether a wetland regulates streamflow and how, at best, monitoring informs this role. With limited long-term monitoring of river flows and wetland storage to determine streamflow regulation roles of a wetlands, classification systems and previous studies can be referred to for identifying streamflow regulation expected from a wetland. According to classification systems, streamflow regulation is variable and specific to a wetland type. Therefore, there are no generalisations for wetlands always regulating streamflow or attenuating or supplementing flows in the same way, all the time, for every event.

*Table 1. Streamflow regulation by wetland type in the HGM classification level (modified from Mbona, 2016)*

| | **Streamflow attenuation** | | **Streamflow supplementation** | |
|---|---|---|---|---|
| | Early season[2] | Late season | Early season | Late season |
| **Floodplain** | ++ | + | 0 | 0 |
| **Channelled valley-bottom** | + | 0 | 0 | 0 |
| **Unchannelled Valley-bottom** | + | + | + | + |
| **Seep with a stream** | + | 0 | + | + |
| **Seep without a stream** | + | 0 | + | + |
| **Depression** | + | + | 0 | 0 |
| **Pan** | + | + | 0 | 0 |

[1] excluding rivers which are listed as an HGM

[2] all streamflow regulatory roles were associated with the wet season

[3] Rating explanation:

0 the function is unlikely to be performed to a significant extent

\+ the function is likely to be present to some degree

++ the function is very likely to be present and often performed to a high level

Concerning the hydrological role of wetlands, streamflow regulation can be affected by factors related to the wetland or the surrounding catchment (Acreman and Holden, 2013). Firstly, factors within (i.e. intrinsic factors) or surrounding the wetland (i.e. extrinsic factors) may affect streamflow regulation. In terms of intrinsic factors, the main intrinsic factor is the wetland storage status. The amount of water retained in a wetland and the remaining deficit required to completely saturate the wetland determines whether incoming flows are attenuated and stored or simply flow through the wetland (Morris and Camino, 2011). By implication, the interplay of water coming into and leaving the wetland affects streamflow regulation. In other words, a large storage deficit during high flows is more likely to lead to streamflow attenuation while during low flows, a large wetland storage deficit is likely to lead less streamflow supplementations as the wetland needs to increase the volume of water it contains. Other intrinsic factors influencing flow regulation includes the wetlands vegetation properties (e.g. density, rooting depth and water use) and physical, hydrological soil properties (e.g. soil textures, depths, infiltration rates, water holding capacities and drainage rates) which jointly affect the wetland's ability to intercept and hold water (Faul *et al*., 2016; Mitsch *et al*., 2009).

In terms of extrinsic factors, streamflow regulation may vary with the wetland's location in the catchment, geomorphology and subsequent groundwater dependence, and climate. The wetland's location in the catchment ultimately determines the amount of water the wetland has access to. Riparian, downstream wetlands have a larger catchment area contributing to the wetland storage which results in the wetland being more likely to receive water from the adjoining rivers, aquifers and upstream land uses. Alternatively, geographically isolated wetlands have a smaller area draining into the wetland and only affect the river flows when the wetland is full and connected to the main river. These factors are reflected in the catchment area draining into the wetland (Lee *et al*., 2018a; Blanchette *et al*., 2022), the land uses and management practices in the surrounding catchment (House *et al*., 2016; Blanchette *et al*., 2019), the connection of the wetland to surrounding or underlying groundwater aquifers (Maherry *et al*., 2017; Melley *et al*., 2017) and cycles of erosion (Pulley *et al*., 2018). From these studies, generally, a riparian wetland with a large contributing area from the surrounding catchment, land uses which are not water intensive and significant groundwater inflows or perched water tables due to a restrictive geological layer underlying the wetland will have a greater wetland storage and impact on catchment streamflow. In addition to this, the wetland's location in the catchment can be related to its topographical setting. According to Savenije (2010), the wetland's topographical setting reveals the dominant runoff flow with wetlands having mostly saturated overland flow. Furthermore, relating altitude to ecological services from wetlands, Chatanga *et al*., (2020) found that high-altitude wetlands have greater impacts on streamflow regulation while low-altitude wetlands promote water purification.

Concerning the climate, wetland storage is responsive to the prevailing climate conditions. Monitoring with a high-density network for a riparian wetland found that changes in rainfall and evapotranspiration are linked to changes in the wetlands surface water or groundwater storage (House *et al*., 2016). Generally, rainfall increases the wetland storage while evapotranspiration decreases the wetland storage. The aforementioned wetland storage variability, wetland type and the wetland storage are examples of intrinsic indicators or factors which determine whether streamflow regulation occurs.

In the context of supplementation and the wetland's storage deficit, the reduced likelihood for streamflow supplementation could be exacerbated with accompanying high evapotranspiration characteristic of the South African climate together with the water use from wetland vegetation and ongoing invasions of alien vegetation (Savage *et al*., 2017; Sieben *et al*., 2021; Scott-Shaw *et al*., 2017). In terms of climate change, although there is uncertainty in how climate change will unfold in Southern Africa, climate change is anticipated to manifest as increasing climate variability (Schulze, 2012; Nehemachena *et al*., 2020) and stressors that will lead to reduced streamflow and reservoir storage (Yamba, 2011; Nhamo *et al*., 2018). In this context, the additional water supply from wetlands is crucial. Additionally, changes in the climate could lead to more streamflow regulation activities by wetlands which have the capacity to do so. As a result, there is institutional and public buy-in at the national (Working for Wetlands, 2005; Turpie *et al*., 2017; Mudavanhu *et al*., 2017), provincial (Bonthuys, 2020) and local level (Mander *et al*., 2010; Nemutamvuni *et al*., 2020) into the value of wetlands for water security.

Secondly, different factors external to the wetland properties and processes may affect streamflow regulation: namely, anthropogenic or natural influences such as climate and human disturbances or interventions (e.g. damming, development and watercourse diversions). Both of the extrinsic and intrinsic factors can be categorised as coming from natural processes or man-made interventions. Cumulatively, these factors affect the water movement and retention in the wetland (viz. the wetland's hydrology).

Thirdly, either of the factors or sources influencing streamflow regulation can be variably active. In other words, various influences (e.g. intrinsic and extrinsic factors from anthropogenic or natural sources) can coexist. Moreover, some influences are more active than others (Bredin *et al*., 2019; WULA in Tanner *et al*., 2019). Altogether, these factors and their interplay affecting streamflow regulation highlights that streamflow regulation is specific to the wetland of interest. In addition to this, contextualising a physical wetland is likely to confirm the general expectations provided in the classification system.

In the next section, a review of wetlands and streamflow regulation in hydrological modelling tools and models are presented.

## 2.3. Modelling wetlands

### 2.3.1. Definition and essential features of simulated wetlands

At the most basic level, simulated wetlands are temporally variable storage units with inflows and outflows (Rahman *et al.*, 2016). A water budget is central to the conceptualisation of physical wetlands in hydrological modelling tools and subsequent models. Figure 2 illustrates a schematic diagram of the wetland storage and water balance. In the wetland's water budget, water retained in the wetland and changing over time is referred to as the wetland's storage. In terms of increasing wetland storage, inflows to the wetland storage may be in the form of rainfall, surface water runoff or river flows and groundwater inputs from surrounding or underlying aquifers. In terms of decreasing wetland storage, outflows may occur as evapotranspiration, surface water runoff and groundwater discharge from the wetland. Concerning the spatiotemporal scale of the simulated wetland, the wetland storage is often solved at the same scale as the modelling tool. For example, temporally, a monthly time step model will simulate wetland storage on a monthly basis. Spatially, a modelling tool which represents land units on a grid, modular or HRU basis will solve the wetland storage at the same scale, respectively.

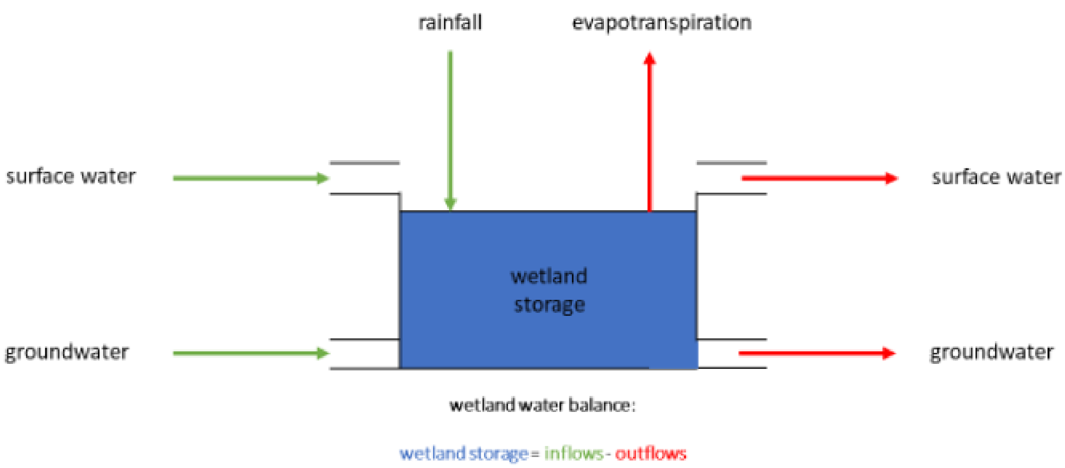



*Figure 2. Common concept of a simulated wetland*

The fundamental definition of simulated wetlands based on the water budget provides the differentiating features of wetland models. Firstly, simulated wetlands may differ in their storage properties. Wetland storage can be estimated at different temporal spatial scales according to the differences in spatial scales and time steps from the plethora of modelling tools available for use. Additionally, simulated wetlands may have different assumed substrate properties. Some simulated wetlands are conceptualised as land units with soil profiles, others assume wetlands are water bodies, and a few are a mix of land and open water (Arnold *et al*., 2012; DHI, 2019). In terms of storage units, some modelling tools have a single reservoir approach where the wetland is one storage unit while other models differentiate the storage into surface and groundwater units (Wolski *et al*., 2006), or soil and water compartments (Arnold et., 2012; Thornton-Dibb *et al*., 2010). Secondly, simulated wetlands may differ by the inflows and outflows included in the water budget. While evapotranspiration is considered in most simulated wetland's water budgets, modelling tools may have different contributing inflows and outflows in terms of surface water and groundwater. Thirdly, simulated wetlands may differ in the algorithms used to calculate each process (i.e. inflows and outflows) in the wetland's water balance. For example, considering rainfall inputs, one model may estimate rainfall inflows to the wetland as net rainfall after evapotranspiration and depending in the wetland's surface area (Pitman and Bailley, 2015). In terms of AET, WRSM-Pitman's wetland model allows AET to persist at an annually repeating rate as long as the wetland is not empty. In the same model, wetland outflows are perceived as a proportion of upstream river inflows which flow through the wetland and spilling only occurs when the wetland is saturated. Alternatively, some models consider rainfall as a weighted average of the wetland's area relative to the entire catchment (Schulze, 1995). In terms of AET, soil and vegetation contribute to the AET based on the soil water content. Outflows from the wetland are dependent on the soil and groundwater storage. Similarly, for other flows in the wetland water balance, different algorithms are possible. In addition to this, each flow may have different parameters in different wetland models. Even when modelling catchment hydrology, this combination of different process algorithms and parameters makes models unique (Clark *et al*., 2009). As a result of so many potential differences from tools and configuration choices in the wetland water budget, there are many simulated wetland models and ways in which the water budget can be conceptualised.

In addition to these differentiating features, simulated wetlands can differ by typology and location in the catchment. At the simplest level, the typology of simulated wetlands differs in terms of the wetland's location in the catchment relative to the river network. A simulated wetland can be a riparian wetland (i.e. within the river floodplain) or geographically isolated wetlands (i.e. outside of the river floodplain) (Figure 3). Notably, all HGM wetland types are either riparian or geographically isolated. According to Rahman *et al*. (2016), riparian wetlands can be further differentiated into in-channel storage (i.e. as part of the river or floodplain with river flowing through the wetland) and off-channel storage (i.e. a wetland which receives part of the main river's flow generally only when the river flow exceeds a bankfull threshold).

Adding a little more detail to the wetland type differentiation, simulated wetlands may be type and tool specific. For example, SWAT's simulated wetlands are developed to represent impoundments, playa lakes, ponds, potholes and prairie wetlands (Arnold *et al*., 2012; Muhammad *et al*., 2019; Evenson *et al*., 2016). Another example is the design of ACRU's wetlands to represent perched wetlands commonly found in drylands and wetland formation on restrictive geological formations (Schulze, 1995; Gray, 2011; Melly *et al*., 2017). This suggests that in some cases, modelling tools have a limited scope for the wetland types incorporated into their model structure.

More intricately, simulated wetland typology may differ by their dependence on the elevation and topography. In complex models with fine spatial scales which maintain the spatial location of features in a catchment, wetlands are defined in depressional areas of the landscape where water can accumulate (DHI, 2019). In simpler models with coarser spatial scales and preservation of the exact location of catchment features, simulated wetlands are not defined by the exact elevation. Rather, the wetland location is implied conceptually with the routing network (for example, placing the wetland as the most downstream component in the catchment to represent a riparian wetland with all upstream inflows entering the wetland) and parameters (for example, in SWAT's wetland HRU, the wetland fraction parameter specifies the proportion of upstream areas draining into the wetland: the greater the value the more a riparian wetland is implied) (Bailey and Pitman, 2015; Arnold *et al*., 2012). Altogether, it is evident that by wetland type, simulated wetlands vary locationally and from tool to tool.

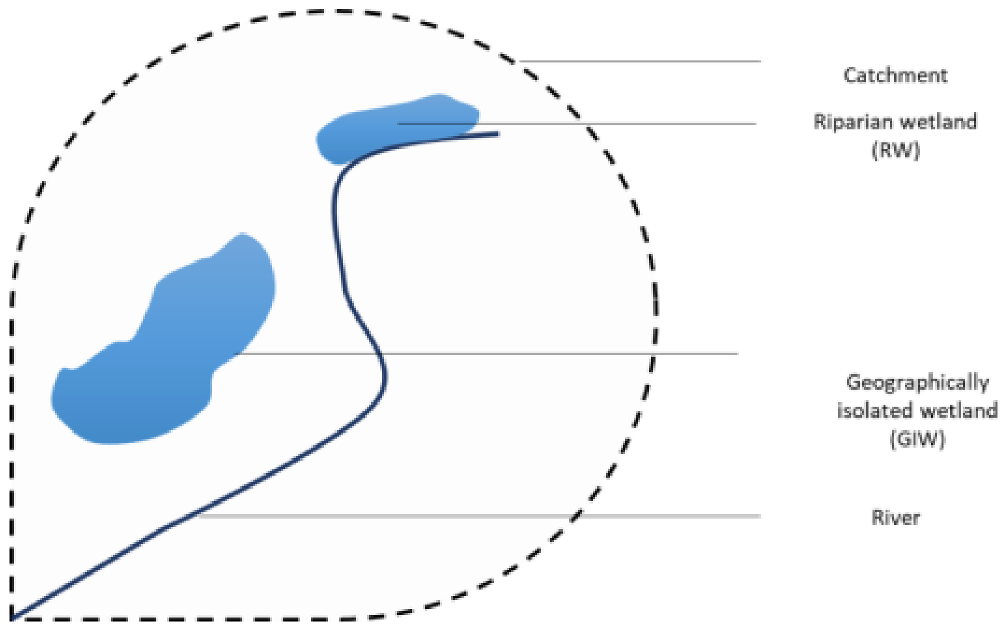



*Figure 3. Conceptual representation of simulated wetland types*

Referring to the model structure, modelling tools differ in their complexity and spatial scale of processes. In the same way that a modelling too dictates the simulated wetlands' temporal and spatial scale, the modelling tool often dictates the simulated wetlands complexity. The complexity of a modelling tool and subsequent wetland models is related to the modelling tool type. In terms of algorithms, the complexity of a model can be high, with physically-based processes (e.g. derived from scientific principles of energy and water fluxes) (Islam, 2011; Brunner and Simmons, 2012), or low, with input-output relationships (Xu *et al*., 2017). Intermediately, a hydrological model's algorithms can be conceptual, using a series of storages with inflows and outflows, storage deficits and parameters loosely based on the physical processes. In terms of spatial scale, hydrological models can be complex in fully-distributed models (capturing the full range of spatial variability at point to grid scales) or simple in lumped models which aggregates the spatial variability and estimates hydrological processes for a whole area using one area or representative value. Intermediately, a hydrological model's spatial complexity may be semi-distributed. Each of these model types have their advantages and disadvantages. Complex models are often acclaimed for capturing spatial variability and properties which influence hydrological responses. However, complex models can be computationally and data intensive. Alternatively, simpler models are less computationally and data intensive but may oversimplify the catchment and hydrological processes.

There is wide discourse on whether increasing model complexity improves process representation and model performance. Some research suggests that more complex models, including physically-based and conceptual model types and fully- to semi-distributed models, improve model performance. For example, Wolski *et al*. (2006) attributed improvements to these models' ability to capture the spatial complexity of the system, topographic controls on the wetland storage and long-term variability of outflows. In addition to this, Makungu and Hughes (2021), model complexity was associated with better descriptions of hydrological processes wetland river-exchanges. Alternatively, there is research suggesting that lumped models are adequate. In this case, it is argued that a system operates as a whole and needs to be investigated as whole instead of its parts (Bloschl *et al*., 2015). Furthermore, other research advocates for neither of the previous extremes. In one case, a review on hydrological models and modelling for water resource decisions in conditions of data scarcity found that neither model types, lumped or distributed, improve model predictions (Devia *et al*., 2015; Tegegne *et al*., 2017). Similarly, a review on wetland modelling suggests that improvements to model predictions from increasing model complexity is limited (Getahun and Demissie, 2018). In this case, model performance and process representations only improve to the level which the data and model structure can support accurately describing the physical wetland. This observation has been supported in a later modelling study where increasing wetland model complexity and spatial representation for the Canadian Prairie pothole improved in the semi-distributed setup but not in the fully-distributed, highly detailed model (Muhammed *et al*., 2019). Therefore, while simple models may miss, or lump together, wetland processes and features that affect the catchment hydrology and wetland storage; at some point, increasing model complexity does not improve catchment or wetland representation in hydrological models.

In terms of how the simulated wetland is incorporated into a modelling tool, the simulated wetland may differ by the spatial scale of the model (Rahman *et al*., 2016). Table 2 summarises the how implementing a simulated wetland differs with spatial scale. In fully-distributed models, wetlands are included in the catchment as spatially explicit areas. In semi-distributed to lumped models, simulated wetlands tend to be incorporated as separate models or sub-routines different from the catchment. The trade-off between both approaches is related to the model's ease of use and realistic representation of spatial variability versus the computational requirements. However, these conceptual and implicit wetland routines are general baselines from which modelling tools construct tool-specific wetland concepts and characteristics.

*Table 2. Simulated wetland concepts by spatial units of the catchment model and trade-offs (after Rahman et al., 2016)*

| Spatial discretisation | Wetland concept | Pros | Cons |
|---|---|---|---|
| **Fully-distributed (grids)** | Explicit topography forms a low-lying area where water can accumulate or create saturated conditions<br>Simultaneous calculation of hydrology at each grid cell | Spatially explicit | Grid size limits wetland area and vice versa<br>Computationally costly as scale becomes finer |
| **Semi-distributed to lumped** | Separate conceptual model<br>Catchment upstream of the wetland and wetland hydrology calculated independently; catchment flows interact with wetland and vice versa | Low computational cost | May not preserve spatial location of wetland<br>Usually designed for one wetland type (alternative can sometimes be implied)<br><br>Spatially limited to conceptual maximum |

Concerning streamflow regulation, simulated wetlands rely on a combination of conceptual or physical processes and software restrictions coupled with subjective modelling choices. In terms of model design, streamflow regulation in hydrological models can be based on factors within the wetland. As wetland storage was central to the wetland water budget, wetland storage is a primary influence on simulating streamflow regulation. This is evident with the common approach for tools and wetland models to regulating wetland inflows and outflows according to the current wetland storage and subsequent deficits or overflows. Essentially, streamflow attenuation is a function of the wetland storage deficit and inflows while supplementation is a function of the wetland storage outflows. Therefore, wetland storage regulation results in streamflow regulation. Volume-based regulations of wetland storage is often described as the fill-and-spill model. The fill-and-spill model differs by wetland type and model. For example, relative to model type, in complex models, the volume-based storage regulation is incorporated using water levels with water moving from highly saturated areas to drier areas, or high-water levels to low water levels (DHI, 2019). Contrastingly, in simpler models, streamflow regulation on a volume-basis is simulated as storage increments when the wetland storage is not full and storage decrements when the wetland storage is full. This approach implies fixed maximum wetland storage capacity and relies on conceptual thresholds and parameters for inflows, water retention or drainage in the wetland storage, and outflows. With conceptual thresholds and parameters, subjective modeller choices are introduced.

Furthermore, simulated streamflow regulation depends on the relationship between the wetland and the surrounding catchment. In terms of model design, the relationship is a function of wetland storage fluxes occurring or responding to changes in water entering or leaving the wetland. In other words, the wetland storage exists within the catchment and not as isolated unit. Therefore, the wetland storage and subsequent streamflow regulation depends on the inflows from the surrounding catchment, and restrictions on the wetland storage permitting outflows. The relationship between the wetland and surrounding catchment can be summarised in three aspects: a hydraulic gradient (Makungu and Hughes, 2021), the direction of water flows through the catchment and the streamflow volumes in the rivers (Leebowitz et al., 2018; Neff et al., 2019; Calhoun et al., 2017). In terms of the hydraulic gradient, wetland exchanges can occur with the surrounding land uses, river flows or groundwater from areas with high volumes to low volumes. Alternatively, wetland exchanges can occur based on the routing network of the catchment and wetland model describing where water flows from and to. This is partly enforced by the modelling tool and the modeller's configuration of the catchment network. Lastly, some wetland models describe inflows into the wetland based on the upstream river flows. One example of this approach is in off-channel, riparian wetlands. Additionally, in model development notes listed in Hughes *et al*. (2013), successfully capturing streamflow attenuation can be achieved by linking wetland outflows to downstream river flows. In this case, low river flows initiate wetland releases, simulating streamflow supplementation, instead of algorithms based on the full wetland storage to initiate wetland outflows. With either factors in the wetland or the wetland's relationship with the surrounding catchment influencing streamflow regulation, both factors and the resulting streamflow regulation are determined by the software's model design and configuration choices by the modeller. Moreover, the combination of these factors needs to complement, or describe, the physical wetland being modelled.

Following this review on simulating wetlands and streamflow regulation, the next section reviews how wetland modelling has been used in different applications and for determining the impact of wetlands.

### 2.3.2. Current scope of wetland modelling studies

Over time, wetland modelling has expanded from replicating observations to being tools facilitating various impact assessments. Early studies were focused on creating wetland models that could be included into existing hydrological models of catchments. As a result, the wetland modelling focused on replicating observations for the wetland water balance. Acreman *et al*. (2007) referred to this as the necessity to capture a wetland's "transfer mechanisms" (viz. how water moves through the wetland under high and low storage and streamflow conditions, the wetland's vegetation, connections to groundwater and dependence on geology). Similar efforts are evident in Zhang *et al*., 2005. More recently, modelling wetlands in hydrological models is applied to water quantity impact assessments and with some examples of water quality monitoring (Yang *et al*., 2016; Weber *et al*., 2021). Concerning water quantity modelling, which is within the scope of this research, wetland modelling has been used in impact assessments from climate change, land cover or use changes, wetland loss and rehabilitation (Fossey and Rosseau, 2016; Rebelo *et al*., 2015; Blanchette *et al*., 2019; Rebelo *et al*., 2022), and for identifying best management practices (Zhu *et al*., 2020; Dash *et al*., 2020; Wolski *et al*., 2006; Mirzaei *et al*., 2021). In addition to this, recent efforts in modelling wetlands focus on improving the existing wetland models in modelling tools (Evenson *et al*., 2016; Rahman *et al*., 2016; Gray, 2011; Pitman and Bailey; 2015; Qi *et al*., 2019; Hughes *et al*., 2013).

In terms of modelling methods, early modelling approaches focusing on several outputs (i.e. evaluating the wetland water balance output from several tools) (Acreman *et al*., 2003; 2007) has shifted towards the sufficiency of one model's output (i.e. perfecting and modifying one a wetland in one tool). In terms of catchment hydrology, model intercomparison studies involved multiple tools and setups (Smith *et al*., 2004; Kollet *et al*., 2017; Garavagalia *et al*., 2017; de Boer-Euser *et al*., 2017; Vansteenkiste *et al*., 2014). However, literature indicates that comparing wetland hydrology primarily focuses on the development of wetlands in one tool. The general trend is to develop the wetland model incorporated in the modelling tool. Thereafter, several efforts and updated works were completed to modify the existing wetland model in a tool or add new wetland models. Examples of this trend can be found in work relating to the Pitman models (WRSM 2000 and updates, Havenga *et al*. 2007; Rayburg and Thoms, 2009; Bailey and Pitman, 2016; Maherry et al., 2016; Makungu and Hughes, 2021; Hughes et al., 2013), ACRU (Smithers, 1991 in Schulze *et al*., 1995; Gray, 2011), SWAT (Arnolds *et al*., 1998, Neitsch *et al*., 2012; Liu *et al*., 2008; Lee *et al*., 2018a, 2018b), and reservoir modelling (Wolski *et al*., 2006).

For modelling methods related to choosing the best model, previous studies use model complexity, risk and scale as indicators of a suitable wetland model. Initially, the proposed framework for wetland model selection and configuration was to consider the complexity by starting with simple models and making the model incrementally complex (or detailed) until model performance stops improving or when all available observation data has been used to validate the model output (Acreman *et al*., 2007). Recent studies still use increasing complexity of a model setup to determine the most suitable setup (Orth *et al*., 2015; Baroni *et al*., 2019). For addressing uncertainty, the level of risk associated with the decision which the model will inform is used to determine how detailed the model could be and how strict the benchmarks for model credibility should be (Acreman *et al*., 2007). Considering that there is no perfect modelling tool or wetland model, the combination of scale and context of use were highlighted as critical determinants for identifying a suitable wetland model (Acreman *et al*., 2007). In this case, compatibility between the spatial scales of the model and physical catchment (e.g. from field to regional) and use cases (e.g. investigating ecological features of the wetland unit or understanding the wetland's functionality and influence on the catchment) guide determining which simulated wetland is suitable.

Currently, modelling wetlands and establishing their credibility relies on model performance assessments. Considering model performance standards, occasionally, the simulated water balance is computed on a long term-basis (i.e. monthly and annually) (House *et al*., 2016; Muhammad *et al*., 2019). More commonly, the catchment streamflow is validated, and it assumed that if the catchment streamflow is replicated well the catchment and wetland models are plausible. However, a recent study found that good model performance for streamflow (viz. the catchment water balance) can be coupled with poor model performance for the wetland storage (viz. the wetland water balance (Evenson *et al*., 2016). As a result, it appears necessary to validate catchment and wetland water balance variables. Considering the wetland unit and modelling approaches relying on the unit, scenario modelling and flow specific analyses are used. The most common scenario setup is the comparison of streamflow from a model setup with and a without a wetland (Fossey and Rosseau, 2016; Wu *et al*, 2019; Bai *et al*., 2021). Alternatively, the long-term model responses are identified (Mandlazi, 2017). In other cases, model development for a particular region, wetland or conditions are investigated (Evenson *et al*., 2016; 2018; Fossey *et al*., 2015; Muhammad *et al*., 2019; Gray, 2011). These approaches also used demonstrate how the impact of wetlands on catchment hydrology is assessed.

Although the impact of wetlands on catchment hydrology are often captured in model performance assessments using catchment streamflow or scenario modelling, there are the potential impact of wetlands on catchment hydrology can be investigated from flow-specific assessments only (e.g. from the model run with a wetland). Within hydrological modelling, time or flow specific assessments related to return periods (Wolski *et al*., 2006; Fossey and Rosseau, 2016; Mandlazi, 2017) or long-term seasonality in the hydrological year and streamflow simulations from dry and wet years have been used to detect the wetlands impact on water availability (Euser *et al*., 2013; Wu *et al*., 2009).

Beyond the scope of this project but noteworthy, approaches combine different modelling tools or use different modelling tools have been used to estimate the impact of wetlands on catchment hydrology. In terms of hybrid models, modelling wetlands has been achieved in hydrological models coupled with hydraulic models (Makungu and Hughes, 2021, Havenga *et al*. 2007; Rayburg and Thoms, 2009). This approach leans on the idea that combined models maximise on their strengths by covering the alternative modelling tool's weakness (Seiller *et al*., 2012; Chomba *et al*., 2021). However, hydraulic models often limit the scale of the study and have high data, intellectual and computational requirements (Ruqayah, 2018; Makungu and Hughes, 2021). In terms of different modelling tools, geospatial modelling considering the extent of wetland inundation is another way of determining the wetland's impact on catchment hydrology (Yeo *et al*., 2019b), especially with advances in estimating wetland storage remotely (De Vries *et al*., 2017).

Wetland modelling studies have generally not applied the same sets of metrics as one another in assessing the modelled impacts of wetlands on the catchment's hydrology. This has resulted in the development of a variety of tailored metrics, but it limits comparability across studies. According to a meta-analysis of studies investigating streamflow regulation services from wetlands, impact metrics are specific to the research question of a study and results in different metrics being used in different studies (Kadykalo and Findlay, 2016). Examples of such metrics are streamflow attenuation estimates at decadal scales and catchment absorption based on runoff responses to rainfall events (Rebelo *et al*., 2015; 2019). The advantage of this variability in impact metrics and several options for estimating impact is that the indicators are specific and contextually relevant. However, the limitation from several options and different metrics used in studies is the inability to directly compare metrics across studies (Kadykalo and Findlay, 2016). This complicates confirming whether a wetland fulfils the streamflow regulation expectations in literature which are also period dependent by season, event type (see Mbona, 2016), and the possibility for a wetland to have variable to no impact on streamflow at different times.

## 2.4. Synthesis and conclusions

The literature and current state of wetland modelling demonstrates that a modeller needs to know if modelling tool or model is suitable for a wetland of interest. This is in response to and as a result of many modelling tools, possible simulated wetland models and physical wetlands with variable streamflow regulation roles (i.e. Influences on water availability). Moreover, simulated wetlands within a modelling tool have a finite scope of wetland concepts in its model structure. In other words, it is likely for the simulated wetland within a modelling tool to be designed to represent the processes of a certain wetland type. Consequently, simulated wetlands need to be evaluated for their suitability to different physical wetlands.

In the review, it was evident that modelling reflects the interaction of wetland characteristics and processes. Jointly, the wetland features enable simulating streamflow regulation. As a result, wetland representation is the collective ability of a model to include a physical wetland’s characteristics, processes and function.

In this context, two research gaps were identified. Firstly, there is a need for standard assessments of wetland characteristics and processes in different models and modelling tools that is wetland-specific. This will enable the comparison of different wetland types and simulated wetlands, and comparison of outputs across studies. Secondly, there is a need to detect streamflow regulation implied in a model, during historical extreme events, before applying the models to impact assessments and future climates. This will contextualise and clarify the wetland’s influence on water availability which can be variable.

In response to these gaps, the following research outlined and conducted wetland model comparisons with and without modelling. Using a case study wetland, in this study wetland representation is relative to a physical wetland. Focusing on the wetland model as an isolated unit, the characteristics and process in the model will be evaluated. Focusing on the wetland in the context of the surrounding catchment, model performance and streamflow regulation in the simulated will be detected. Since hydrological modelling is an intensive task with applications in many water resource decisions, it is hoped that an efficient, standardised and baseline methodology to give wetlands more priority in the model selection process, in academic and industry settings with resource and time constraints, will be outlined. Cumulatively, the research is important for determining whether different models setup for the same wetland and climate results in the same model performance and impacts on streamflow. In addition to this, the research will provide evidence for whether a suitable wetland model (conceptually) correlates with good model performance. This will highlight whether wetland models can be effectively compared with and without modelling. Altogether, these efforts may promote wetland-inclusive modelling and model development for the case study wetland.

The next chapter presents a method to assess simulated wetlands relative to physical wetlands and apply it to a case study wetland.

# Chapter 3: Comparing wetland models

## 3.1. Introduction

### 3.1.1. Background information

Various physical wetlands can be modelled differently in several modelling tools (i.e. software options) and potential models (i.e. configuration choices within the tool and from modeller choices). Modelling tools may have different approaches for representing a wetland. Chapter 2 demonstrated how simulated wetland differences could arise from the surrounding catchment, processes within the wetland or the interactions between the wetland and the catchment. The diversity in simulated wetlands increases with the large number of modelling tools available for use and different setups that a modelling tool can assume to represent a physical wetland. There is a need to systematically define how a wetland routine represents the processes in a specific wetland and the assessment should allow for comparisons across tools.

The standard method for establishing model credibility is with model performance assessments. This relies on the model's ability to replicate observed datasets. Generally, model performance has a catchment-scale focus with streamflow validations (Juniati *et al*., 2018; Pool *et al*., 2018). In this way, the existing solutions for characterising wetland models do not focus on wetland representation since most approaches focus on modelling the catchment-wetland complex and replicating observed streamflow. This may be a limitation to selecting and configuring a wetland model because observed streamflow is not widely available and monitoring points are declining (Pitman, 2011; Wessels and Rosseboom, 2009). In addition to this, streamflow is highly variable (Okello *et al*., 2015) and subject to measurement errors (Beven, 2019). Therefore, currently and conventionally, the wetland water balance and variables are not prioritised in model performance assessments.

However, there are upcoming examples of validating wetland models in modelling studies from a model development perspective. In terms of considering variables from the wetland water balance, recent studies have added wetland-scale model performance assessments by validating wetland storage (Evenson *et al*., 2016) or evapotranspiration (Tanner *et al*., 2019). A few studies have considered modifying the wetland algorithms to include different hydrological processes and flow pathways (Evenson *et al*., 2018; Rahman *et al*., 2016) or the wetland's configuration within the catchment (Muhammad *et al*., 2019). Qualitatively, summaries of wetland concepts have been presented. Maherry *et al.* (2017) reviewed wetland processes and water balances in modelling tools and developed conceptual flow models from the classification of wetlands as HGM units to guide the configuration of a simulated wetland. Multi-criteria analysis, not to be confused with multicriteria model calibration, is a popular choice for guiding hydrological decisions with applications in modelling water quality (Payraudeau *et al*., 2012) and mapping groundwater zones (Al-Ruzouq *et al*., 2019), flood risk areas (Papaioannou *et al*., 2015), water surplus and deficit areas and water quality deterioration (Chung and Lee, 2009). Where multiple runs are required for multicriteria model calibration, the computational and time resources put into this approach can be avoided with the sense-checking assessments of the model's algorithms compared to what is understood about the wetland system in reality. However, few examples of sense-checking model structures for a particular case have been found for hydrological modelling for water quantity estimates. In terms of characterising physical wetlands at the wetland-scale, multi-criteria analysis has been used for the assessment of wetland ecosystem services, integrity and flow regulation with examples of such procedures outlined in the WET-Ecoservices (Kotze *et al*., 2007; 2019), WET-Health (Kotze and Ellery, 2009) and a qualitative review of reported wetland functions (Bullock and Acreman, 2003), respectively. These options are useful for their intended purposes and contexts; and highlight how the comparison of physical and simulated wetlands are yet to be considered.

### 3.1.2. Method development and justification

One way forward is to combine the elements of existing solutions. Multi-criteria analysis can be used for categorically rating simulated wetland features relative to the physical wetland. Classification systems can serve as 'observed data' defining the wetland properties and behaviour. Since the fundamental properties of simulated wetlands have been identified, these wetland features could serve as the criteria for the comparison. This is convenient considering the lack of observed data to facilitate model performance assessments for most wetlands.

The purpose of the modelling can guide the selection of wetland or catchment model scales to assess in the comparison. The spatial scale of modelling has been linked to the type of information the model output generates. Hydrological modelling tools have a range of spatial scales for which they are valid. A modelling tool is usually distributed towards one end of the range of possible scales from field to global scales. According to Armstrong (2007), larger scales with the wetland included as a functional unit within provides information related to water availability that is necessary for water management while smaller scales focusing on the wetland provides information about the wetland functions, ecology and biodiversity. Extrapolating this logic to catchment-scale modelling, modelling wetlands at the catchment-scale can focus on the catchment with the wetland or the wetland unit. Moreover, the catchment-scale yields information about the wetland's impact on water availability (Armstrong, 2007). Alternatively, wetland-scale assessments provide information about the individual wetland's properties and rules governing its function.

The procedure for comparing simulated wetlands can be practical and relevant by considering the user and data requirements and constraints. Acreman *et al*., (2007) found that academics and practitioners have different approaches to research problems. Practitioners are often required to complete and repeat assessments at many sites within a short time frame which generates risk-based information that leads to implementing water-related decisions or advice to stakeholders. Academics usually have long-term, detailed research in a smaller number of sites and approach the research question with high levels of complexity and process understanding. The outputs of this research are usually channelled towards science innovation and journal publications. Therefore, practitioners need quick and repeatable processes while academics often have access to site-specific datasets and need research developments. This means that the qualitative assessment of wetland representation should be fast and standardised in a way that allows the method to be repeated with different tools and sites. Comparing simulated wetlands in any setting synergises site-specific information and efficiency.

Data requirements for comparing wetlands can be kept low by using widely available information and minimising in-situ data collection. Physical wetlands can be described from previous research on the wetland of interest or literature on similar wetland types. Classifications systems are another source of information for physical wetlands. Simulated wetlands can be described from the modelling tool's documentation. Using these resources minimises the data requirements of the comparative assessment.

The differentiating features of simulated wetlands can serve as criteria for the defining wetland representation. There are four defining categories of simulated wetlands: the wetland typology, storage media, storage regulation and water balance components (see section 2.3.1 in the Literature Review). These criteria are compatible with the information provided in the classification of wetlands as HGM units (Ollis *et al*., 2015; Mbona, 2016).

Altogether, these elements of efficiency in the procedure and relevance in the content provide the foundations for estimating wetland representation at the wetland-scale based on the wetland characteristics and processes.

Although the proposed method does not consider the model's predictions of flow regulation which would require assessing the model output, the method is useful for giving an unbiased view of one or more modelling tool capabilities. Standardised criteria make the method repeatable. The multi-criteria analysis is specific enough to relate to one wetland and general enough to use on different wetland types and simulated wetlands (e.g. tools or models). The repeatable method also lends itself to knowledge accumulation. A committee of international hydrological researchers conceded that hydrological advancements in the next decade are best supported by knowledge accumulation (Bloschl *et al*., 2019). Unlike fragmented knowledge, knowledge accumulation is believed to generate hydrological principles within a network of related fields and applications which is often how hydrology persists in reality. The qualitative assessment alleviates the immediate need for quantitative modelling to establish model credibility. This reduces the time required to determine whether the simulated wetland is a suitable approximation of the physical wetland.

In addition to this, the method has a clear distinction between model expectations from the physical wetland it needs to simulate and model capabilities reflected in how the model can approximate the physical wetland. In this case, the credibility of the simulated wetland is based on the wetland routine alone which has less noise than the catchment-wetland complex and is independent of observational datasets. Model uncertainty (i.e. how the model represents and simplifies physical processes which are not fully understood) and equifinality (i.e. the ability of a model to generate acceptable results from different setups) has been a growing concern in the modelling community (Beven, 2018; 2019). This is sometimes referred to as models giving the right answers for the wrong reasons (Kirchner, 2006). Focusing on the wetland routine investigates how the model approximates reality (e.g. the model structure) and is a step towards modelling results that are the correct answers for the correct reasons.

### 3.1.3. Aim

The aim of this chapter is to compare wetland representation from wetland concepts in modelling tools and from the understanding of a real wetland. The analysis presents a method to do so and the suitability of selected simulated wetlands for a case study wetland.

## 3.2. Methods

### 3.2.1. Wetland locality

The case study wetland is located in the Upper Kromme River catchment (K90A quaternary catchment), Eastern Cape province, South Africa (Figure 4). It has been estimated that, due to land cover change, management, and alien plant invasion, the Kromme's palmiet (*Prionium serratum*) wetland coverage has reduced by approximately 83 % from 10 $km^2$ in pre-development to the current 1.7 $km^2$ (Cornelius *et al*., 2019; Rebelo, 2012; Rebelo *et al*., 2015; Figure 4).

The case study wetland is on private land within the Krugersland farm and constricts at the R62 bridge. After the R62 bridge, the wetland area expands again into palmiet on the Kompanjiesdrift farm. The Krugersland and Kompanjiesdrift wetlands are both relatively pristine and mostly vegetated with palmiet. However, further downstream there is significant encroachment of black wattle (*Acacia mearnsii*) and perennial reeds (*Phragmites australia*) (Figure 5). The qualitative analysis focused on the uppermost palmiet wetland (i.e. the Krugersland wetland).

### 3.2.2. Wetland type and hydrodynamics

The wetland is classified as an unchannelled valley-bottom wetland within the hydrogeomorphic (HGM) unit framework (Ollis *et al*., 2013; Figure 6). This classification level describes the wetland's local setting (i.e. landform shape and position), hydrological characteristics (i.e. water movement into, through and out of the wetland) and hydrodynamics (i.e. the direction and strength of flow through the wetland). Similar to the classification's generalisations, the case study wetland is located on the valley floor of the main river network, riparian and all upstream river flows are inflows into the wetland. There is no distinct river channel in the wetland. River inflows transform into diffuse surface and subsurface flows through the wetland's variably saturated land mass. The classification attributes the wetland formation to the changing gradient of the riverbed. Previous studies in the case study site have linked the changing gradient to gully erosion and cycles of cutting-and-filling (Job, 2014; Lagesse, 2017; Mc Namara, 2018). Outflows from this wetland type are usually diffuse or surface flows, in this case concentrating into a downstream river channel.

These wetlands tend to have large storage capacities and infiltration, together with significant AET water losses (Ollis *et al*., 2013; Tanner *et al*., 2019).

According to the classification system's description of unchanneled valley-bottom wetlands and confirmed by local monitoring, groundwater is a significant input into the Krugersland wetland. Groundwater inflows and subsurface outflows sustains and regulates the wetland's storage. Tanner *et al*. (2019) presented a geologically-dependent concept of the catchment and wetland following detailed monitoring of the Krugersland and Kompanjiesdrift wetlands. Firstly, the study results showed that a large proportion of groundwater reaches the wetland via upstream river flows. Aquifer outflows in the highlands flow over steep, rocky slopes towards the river. The wetland receives river flows from the main channel and tributary inflows between mountain ranges to the North and South of the valley (and wetland) (Figure 5). Notable subsurface flows from tributary catchments feeding the valley and wetland via alluvial fans have been observed (Tanner *et al.*, 2019).

Secondly, alluvial fans bordering the wetlands contribute large volumes of subsurface flows to the wetland. Thirdly, the wetland formed on impervious bedrock and has minimal to no contact with underlying aquifers. This classifies the wetland as perched according to the works of Melley *et al*. (2017). The wetland storage profile grades from peat accumulations to underlying sand with the depth to bedrock ranging from 6.0 – 8.0 m (Grundling *et al*., 2017; Pulley *et al*., 2018). Figure 7 presents the average depth of peat and sand from sixteen transects (Pulley *et al*., 2018; Lagesse, 2017) and the main inflows and outflows from the case study wetland.

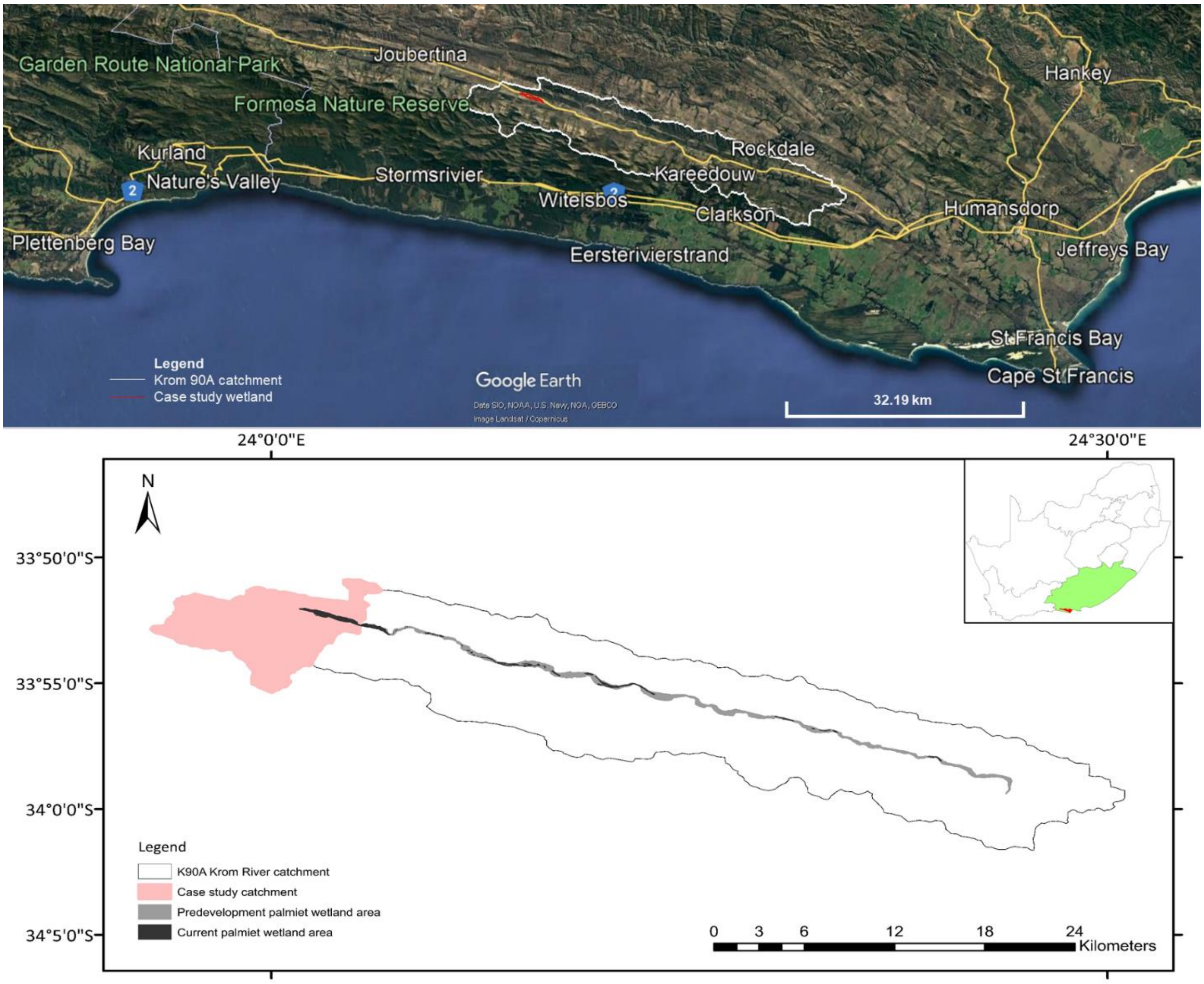



*Figure 4. Upper K90A catchment and case study wetland locality*

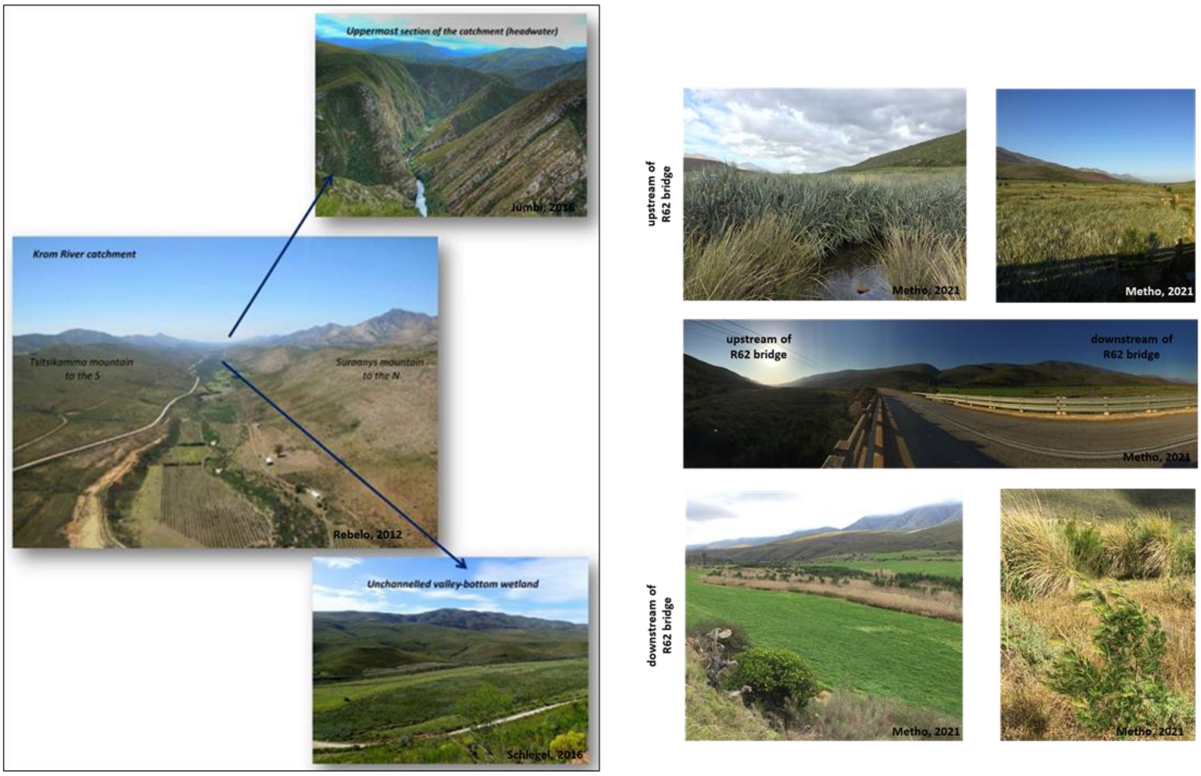



*Figure 5. Palmiet wetland locality (left) and vegetation cover up- and downstream of the R62 bridge (right)*

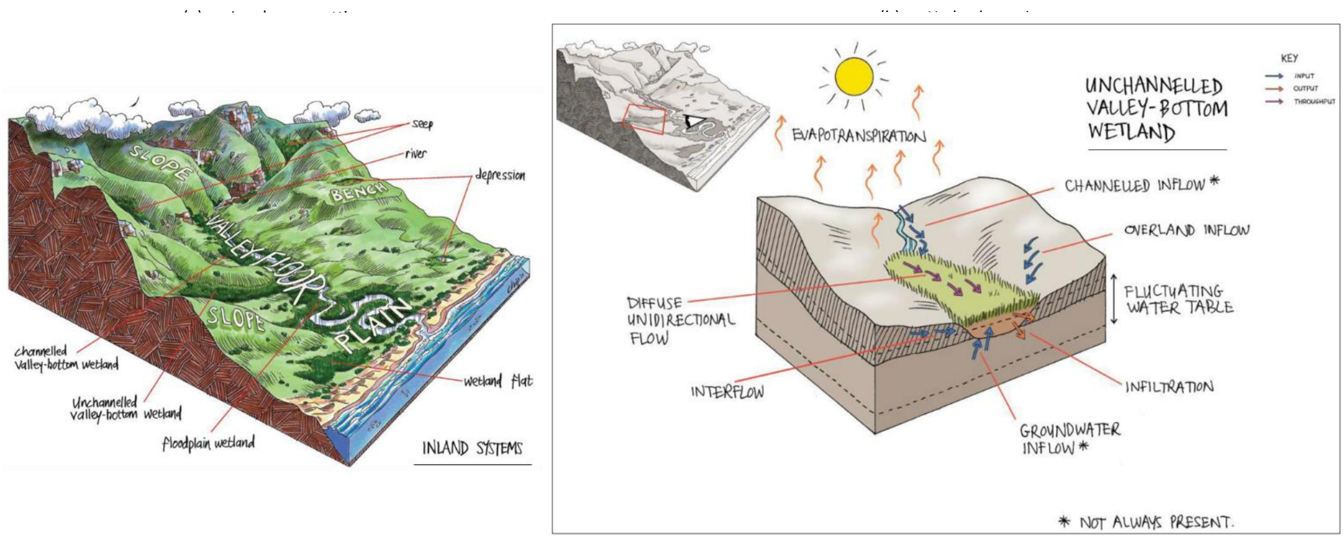



*Figure 6. HGM classification of the unchannelled valley-bottom wetland (modified from Ollis et al., 2013)*

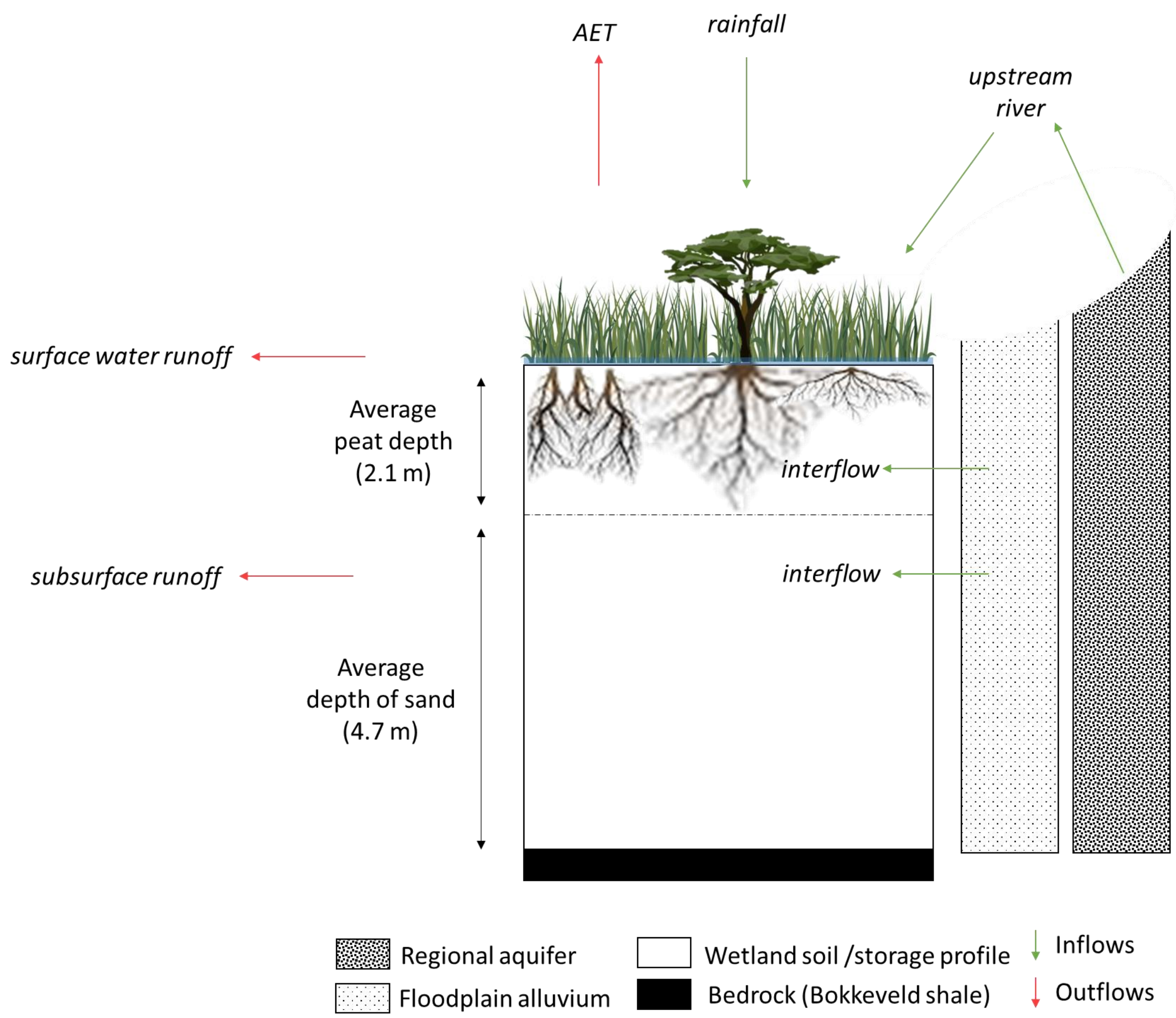



*Figure 7. Schematic diagram of case study wetland inflows and outflows*

### 3.2.3. Modelling tool selection

Modelling tools from which to assess simulated wetlands were selected based on their extensive use in South Africa, availability, and potential to assist research interests.

Two locally developed modelling tools were selected. ACRU4 (Schulze, 1995 and updates) and WRSM-Pitman (Bailey and Pitman, 2016) are freely available tools. ACRU4 (hereafter referred to as ACRU) describes its catchment as hydrological response units (viz. land units with similar properties and responses to rainfall) grouped at the subcatchment scale. WRSM-Pitman is a lumped rainfall-runoff model with the catchment units, and wetlands, expressed as modules within a river network. Both modelling tools have been extensively used and verified. Their model concepts and data requirements are tailored to South Africa's environmental conditions and data availability. Assessing these modelling tools supports the previous and continued use and development of these tools.

SWAT (Arnold *et al*., 2012) and MIKE SHE (Refsgaard and Storm, 1995a; Refsgaard *et al*., 1998; 2010; DHI, 2019) were selected from the international options of modelling tools. SWAT is a freely available tool with the option to spatially aggregate land units into HRUs. The version of SWAT used in this assessment was ArcSWAT2012 (Neitsch *et al*., 2011). Similar to ACRU, the SWAT modelling tool is semi-distributed. The modelling tool comes with a parameter and climate data base that has the potential to alleviate data availability constraints experienced in South Africa.

MIKE SHE is a fully gridded model: the catchment is discretised into spatially explicit grids resembling the actual surface properties of the area in each grid. Although it requires a license to operate, it offers integrated surface and groundwater modelling. Hydrological models are often limited by simple groundwater routines (Acreman, 2007) and integrated modelling of surface and groundwater may become increasingly relevant with conjunctive water use (i.e. using groundwater to supplement water resources) (Hedden and Cilliers, 2014; Hedden; 2016). Several wetland modelling studies have also shown growing interest in groundwater representation (Tanner *et al*., 2019; Maherry *et al*., 2017; Mandlazi, 2017). See section **Error! Reference source not found.** for a description of model types by complexity of the spatial representation and algorithms. In the models explored, MIKE SHE computes the runoff from the land units while Hydro River simulates the channel flow.

## 3.2.4. Comparative analysis

### 3.2.4.1. Criteria

The qualitative analysis compared the underlying concepts in the simulated wetlands to the physical wetland based on definitive criteria. The analyses considered simulated wetlands setup for the Kromme wetland not the range of configuration possibilities offered by each modelling tool. The simulated wetlands were assessed in terms of the fixed representation options from the tool and the user choices that suit the case study wetland.

The concepts for the simulated and physical wetlands were outlined according to the following criteria:

- Basic premises of the wetland concept
- Wetland's dependence on topography (i.e. viewed in terms of the wetland models dependence on the landscape position, catchment routing network and properties to influence the wetland's processes)
- Wetland typology relative to the river network (i.e. riparian or geographically isolated)
- Wetland type the wetland model within the selected tool was designed for
- Water balance (i.e. components and flow pathways in the water balance inflows and outflows)
- Wetland water storage media (i.e. land mass, water body or hybrid)
- Storage regulation (i.e. processes and thresholds governing the wetland inflows and outflows)

### 3.2.4.2. Compatibility score and verdict

A scoring system was used to rate the applicability of the model wetland representation for the Kromme wetland as a numerical value.  A score of 0 to 2 was assigned to each simulated wetland for each criterion. 0 represented no match, 1 represented some compatibility (2 or more "misses"), and 2 represents full compatibility (only 1 or no "misses"). The rating system attempts to translate the wetland concepts side-by-side into a measure of compatibility. It acted as a tool to reduce what could be too detailed, technical or confusing into a final statement about the "hits-and-misses" of the wetland representation. A compatibility verdict was calculated as the total score across all criteria.

## 3.3. Results

### 3.3.1. Wetland model options

The modelling tools offer several ways of representing a wetland, structurally (i.e. inherent in the model design) and by users choice with different parameterisations and wetland or catchment configurations.

#### 3.3.1.1. ACRU

ACRU offers four simulated wetlands options: two explicit wetland components and two setups to imply alternative wetland types. The main, explicit simulated wetland routines are the riparian zone HRU and wetland HRU (Thornton-Dibb *et al*., 2010; Gray, 2011). Both wetland components are inherently riparian wetlands. Other implicit representations can be inferred, namely, a wetland with free standing water by adding a dam downstream of the wetland and at the catchment outlet (Thornton-Dibb *et al*., 2010); or a geographically isolated wetlands with subcatchment divisions into areas contributing to the wetland and areas contributing to the river (Gray, 2011). The qualitative analysis assessed the wetland HRU and the riparian zone HRU within one subcatchment.

The wetland components in ACRU have the same storage profile and different inflow pathways. Both HRUs have a soil profile divided into the topsoil and subsoil. A groundwater storage zone underlies the soil profile. Both the riparian and wetland HRUs receive water on their surfaces as overflow from an associated river channel unit. User-defined channel capacity thresholds initiates spilling from the river into the wetland. The riparian zone HRU can proportion upstream groundwater outflows into runoff added to the wetland subsoil (i.e. as subsurface runoff) and runoff inflows received as river discharge (Figure 8). The wetland HRU receives all surface water runoff and groundwater outflows from the upstream land uses as river discharge (Figure 9). Both wetland components have the same outflows: evapotranspiration from water in the soil profile, surface runoff, delayed surface runoff and groundwater outflows.

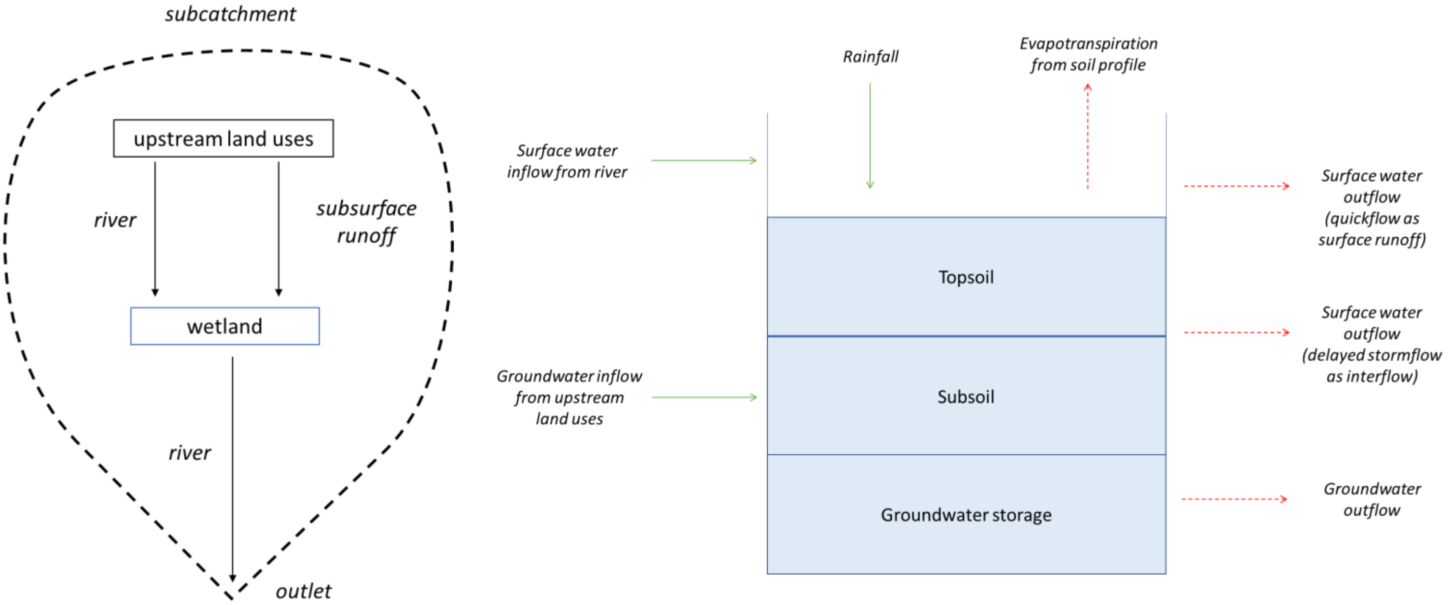



*Figure 8. Schematic diagram of ACRU's riparian zone HRU in the catchment (left) and wetland storage fluxes (right)*

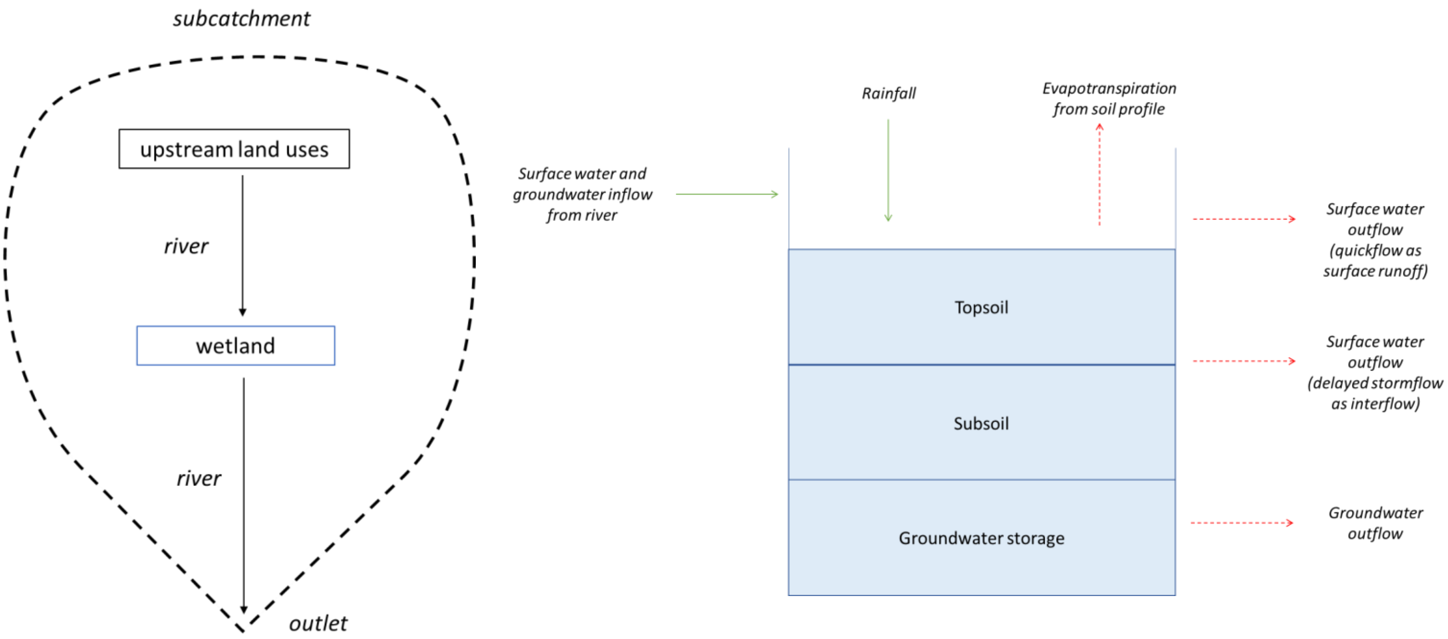



*Figure 9. Schematic diagram of ACRU's wetland HRU in the catchment (left) and wetland storage fluxes (right)*

#### 3.3.1.2. MIKE SHE coupled with MIKE Hydro River

MIKE SHE-Hydro River (hereafter referred to as MS-HR) does not have a dedicated wetland unit or routine, but land and river conditions for specific grid cells can be configured to predict the saturated conditions of a wetland area (DHI, 2019). Saturated conditions can be implied by depressional topography, bathymetry, high surface roughness promoting detention storage, large soil water storage capacity and low conductivity. Upstream land uses are spatially explicit and defined within grid cells (i.e. the spatial extent and location of land uses in the real catchment are preserved in the model). MIKE SHE models the subsurface as an unsaturated and saturated zones. A semi-distributed, conceptual setup of subcatchments or fully-distributed, physical setup of grid cells representing the catchment can be chosen by the modeller. The differences between the two setups are the groundwater representation and runoff components.

The conceptual model has a baseflow reservoir for the whole catchment and groundwater inflows into the wetland only includes capillary rise. In the detailed and distributed setup, groundwater cells are specified in each grid, representing the local aquifer conditions, together with the associated groundwater flows. Interflow is simulated from the saturated zone. In either setup, surface runoff is generated from the unsaturated zone according to the hydraulic (i.e. detention storage or water level from cell to cell) and topographical (i.e. downslope) gradient. Water can move between the saturated and unsaturated zones representing recharge and capillary rise. In addition, the thicknesses of these zones are dynamic such that the water table can be modelled as rising to the land surface in places, leaving no unsaturated zone and allowing groundwater seepage to contribute to surface flow. Water in the saturated zone, unsaturated zone and detention storage available for evapotranspiration.

However, the implied wetland conditions have different channel overflow representations and calculation processes. There are two main approaches for representing wetlands receiving channel overflow which can be differentiated by the wetland grid complexity. The first method employs a fast calculation scheme where water (i.e. all upstream river flows) moves downstream through a simple, general floodplain topography (hereafter referred to as the flood zone wetland option) (Figure 10). The second method uses a slower calculation scheme where water (viz. all upstream river flows) can move downstream and spread across the floodplain to the surrounding land (i.e. through a more complex topography and wider floodplain area). This option uses the overbank spilling routine (Figure 11). Other simulated wetland options can be created from these methods for generating runoff mechanisms and channel overflow representations. Both channel overflow representations were considered in the analysis: the flood zone was paired with the simple saturated zone and the overbank spilling was combined with the fully-distributed, complex saturated zone.

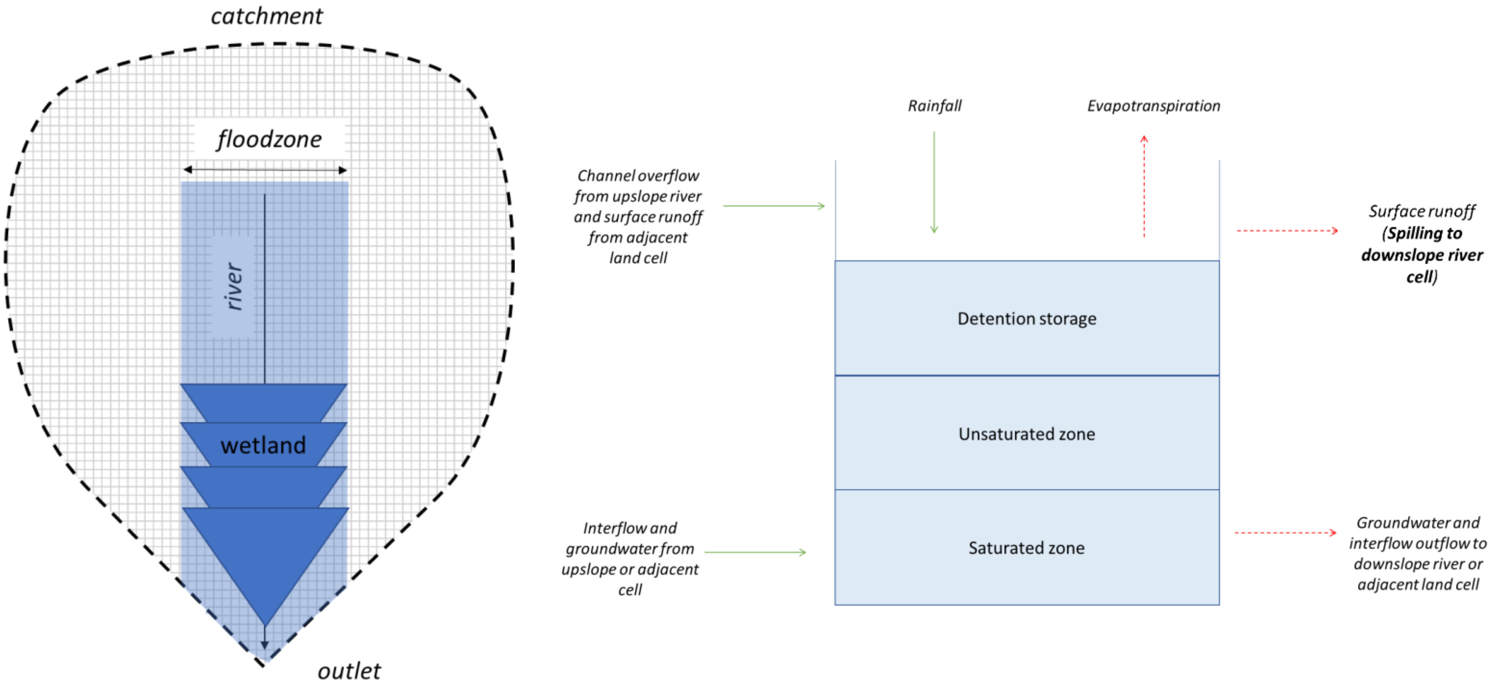



*Figure 10. Schematic diagram of wetland conditions in MS-HR with channel overflow restricted to the flood zone In the catchment (left) and wetland storage fluxes (right)*

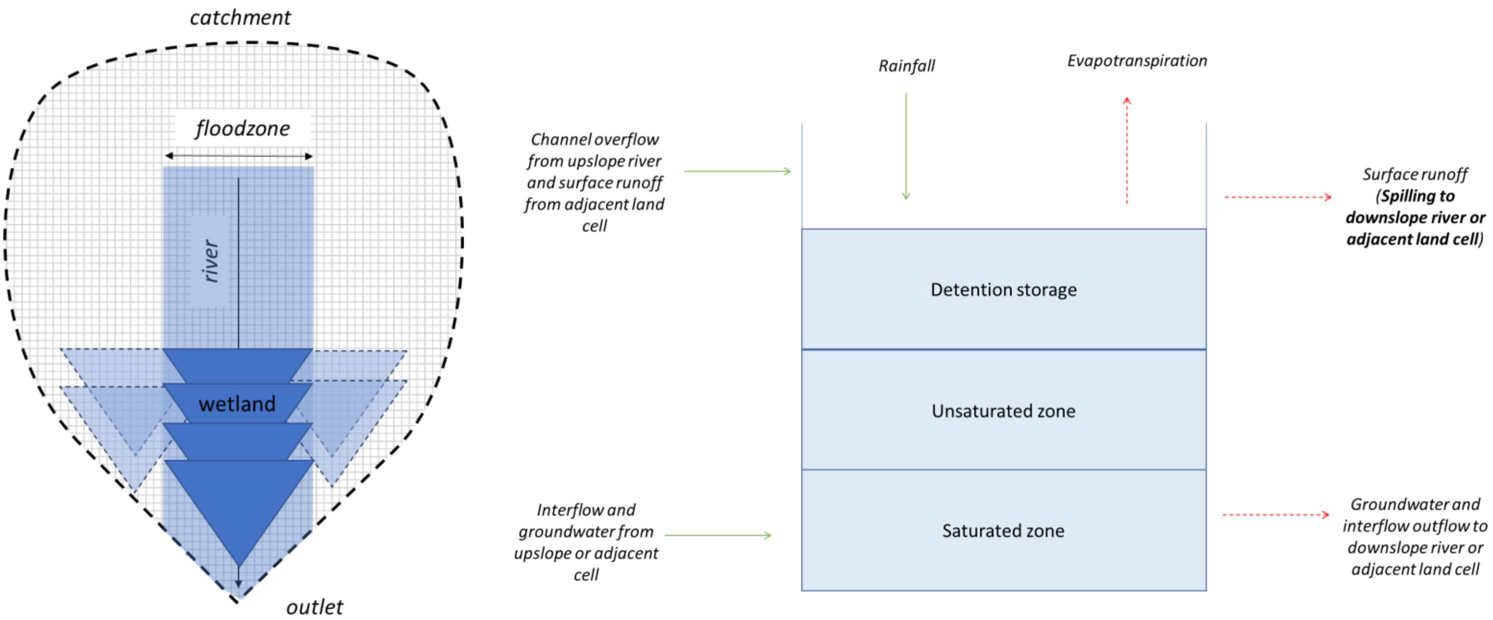



*Figure 11. Schematic diagram of wetland conditions in MS-HR with unrestricted channel overflow In the catchment (left) and wetland storage fluxes (right)*

#### 3.3.1.3. WRSM-Pitman

WRSM-Pitman has two simulated wetland options: a simple and comprehensive wetland module. The latter was assessed in this study and is an update to the simple wetland which replicates a reservoir from older versions of the modelling tool. The qualitative analysis assessed the comprehensive wetland module (Pitman and Bailey, 2015).

The comprehensive wetland module conceptualises the wetland as an open water body (Figure 12; Pitman and Bailey, 2015). Inflows into the wetland depend on the upstream land uses which feed into the river segment upstream of the wetland. The river segment contains the upstream surface and groundwater runoff. Wetland inflows are regulated by the river discharge upstream of the wetland and the channel capacity threshold to initiate overbank spilling. The modeller specifies these discharge rates. A riparian wetland, defined as in-channel storage within the WRSM-Pitman tool, can be simulated by setting the channel capacity to 0 $m^3$/month which allows all river discharge to be an inflow into the wetland. When overbank spilling is initiated, the inflows into the wetland can be routed to the wetland storage or flow through the wetland. The maximum wetland storage volume is required for setting up a comprehensive wetland module. Wetland outflows are computed when the wetland storage is full. The runoff outflow rates are defined by the modeller and regulated by the proportion of wetland storage exceeding the maximum volume. The wetland runoff outflows are routed to the downstream river segment.

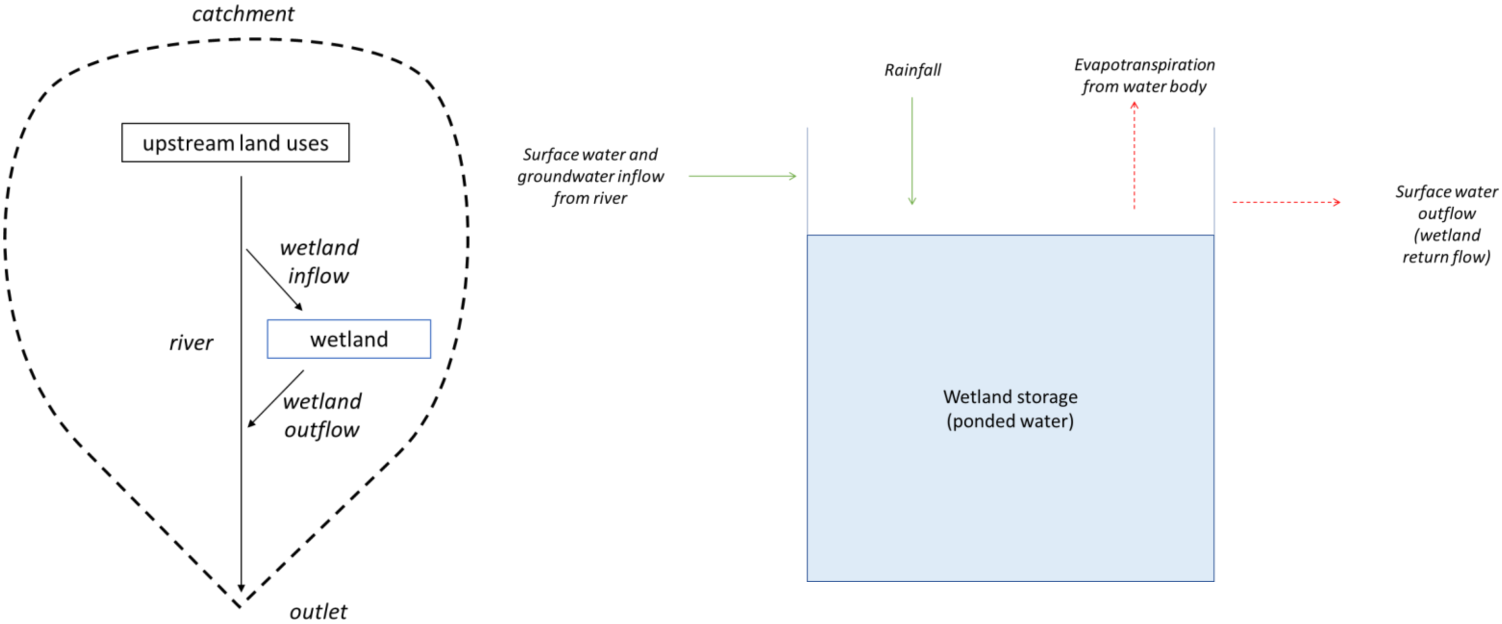



*Figure 12. Schematic diagram of WRSM-Pitman's comprehensive wetland in the catchment (left) and wetland storage fluxes (right)*

#### 3.3.1.4. SWAT

The SWAT modelling tool offered four waterbody options that could potentially be used to represent a wetland: a reservoir, pond, wetland and pothole (Arnold *et al*., 2012; Neitsch *et al*., 2011). A SWAT reservoir could imply a primarily surface water storage wetland and requires a set of regulatory spillway volumes that are not applicable or available for the case study wetland. The case study wetland is not an open water body (i.e. lake or pond). Therefore, only the wetland and pothole simulated wetlands were assessed. Additionally, these options are more likely to be intuitively selected for modelling a wetland from according to their terminology.

Simulated wetlands in SWAT receive surface runoff, interflow (or lateral runoff) and groundwater from the upstream land uses (Neitsch *et al*., 2011). The catchment-wetland complex and water balance for the wetland HRU and pothole HRU are illustrated in Figure 13 and Figure 14, respectively.

The inflow in the wetland HRU is in proportion to the user-specified subcatchment area draining into the wetland and depends on the current surface area of the wetland in the subcatchment. Inflows into the pothole HRU are from the user-selected HRUs which flow into the wetland and in proportion to the user-specified HRU area draining into the wetland. The pothole HRU was developed for agricultural land uses and allows irrigation as an additional input into the component.

The wetland and pothole HRUs generate runoff outflows when the normal wetland surface area and the storage capapcity are exceeded, respectively. The wetland HRU routes outflows to the main channel and the pothole HRU routes wetland outflows to the lowest elevation point within its HRU similar to water retention in playa lakes and artificial impoundments (Figure 14). The pothole outflows indirectly and eventually affect the river flows with subsurface runoff to the main channel and recharge from the soil profile of the pothole HRU. Seepage outflows are regulated by the saturated hydraulic conductivity in the wetland HRU and the soil water content in the pothole HRU. The wetland HRU routes seepage to the river and the pothole HRU directs seepage outflows to the underlying soil profile when the soil water content is less than 50 % of the field capacity (i.e. relatively dry).

The wetland storage profiles and subsequent AET routines of the SWAT wetlands are different. The wetland HRU estimates AET as water losses from an open water body with the wetland storage conceptualised as an open water body. The pothole HRU adjusts the open water AET by the shading of the vegetation using time varying leaf area indices.

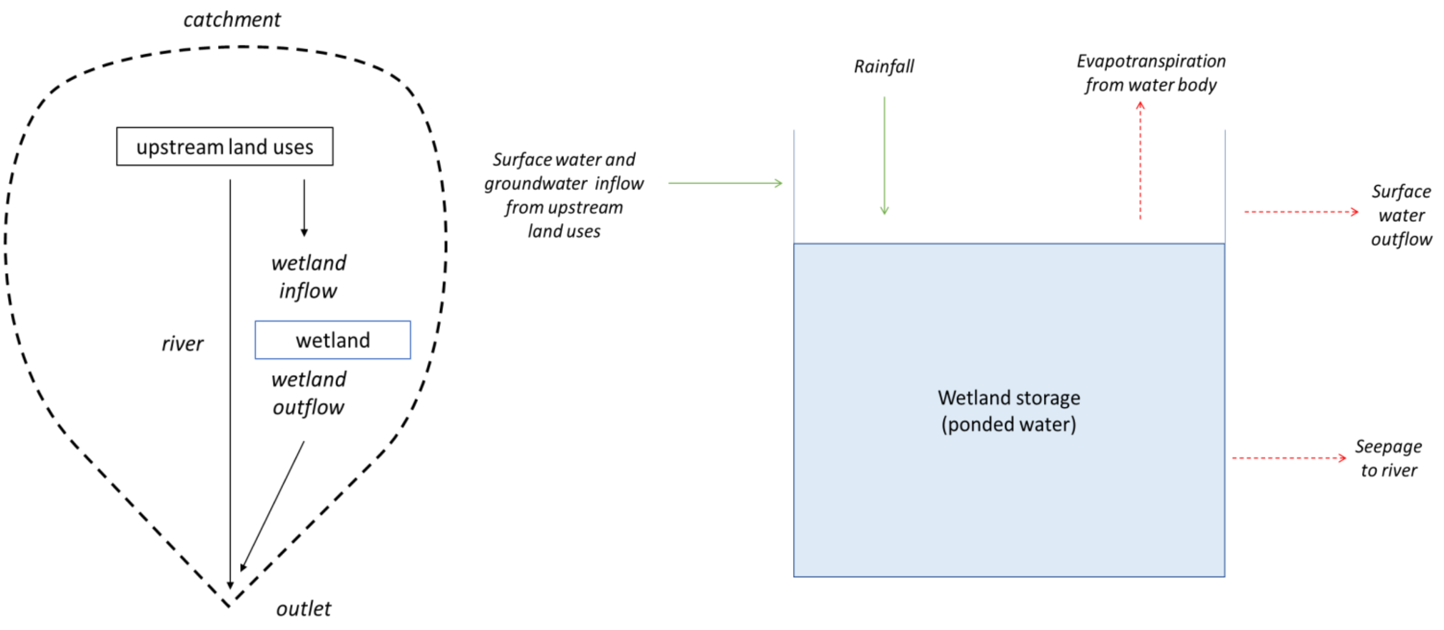



Figure 13. Schematic diagram of SWAT's wetland HRU in the catchment (left) and wetland storage fluxes (right)

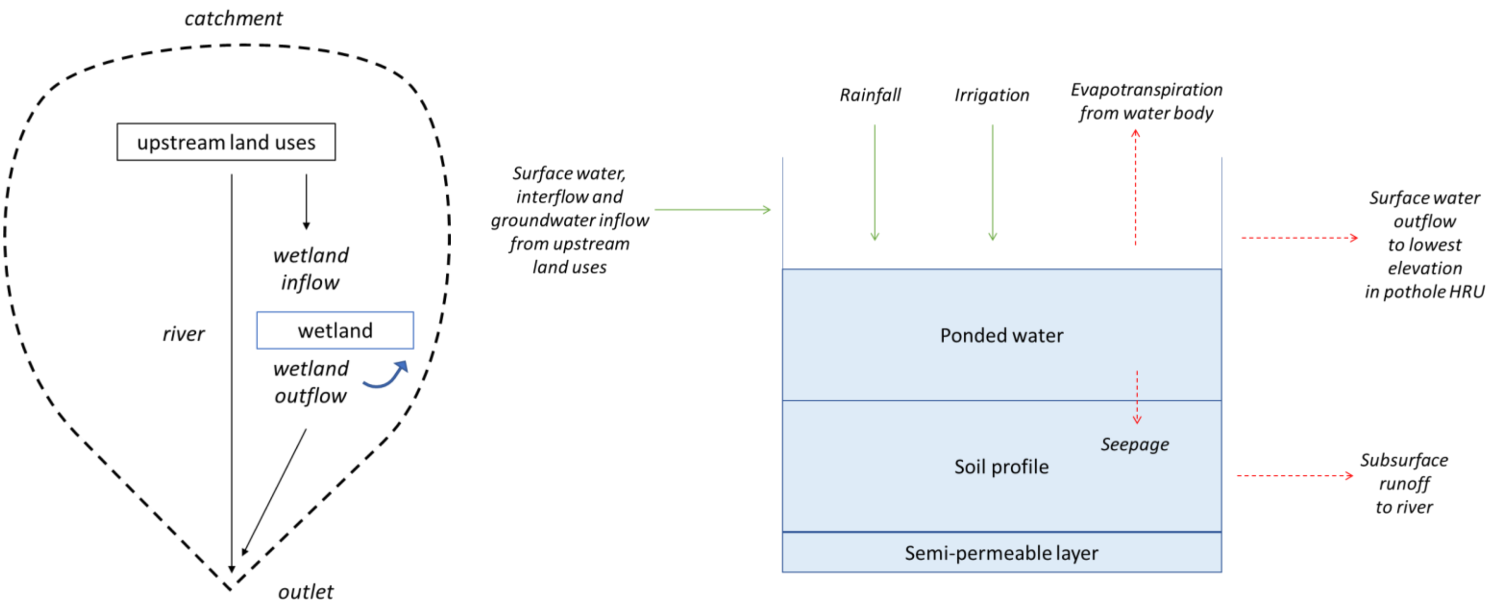



Figure 14. Schematic diagram of SWAT's pothole HRU in the catchment (left) and wetland storage fluxes (right)

### 3.3.2. Compatibility

All simulated wetlands show potential to simulate wetland inflows but have different, implicit and conceptual approximations of how the process occurs. Wetland storage was fundamentally regulated by the deficit storage volume. Table 3 presents the Kromme's unchannelled valley-bottom wetland features and potential wetland models from the options described in section 3.3.1. Table 4 presents the compatibility verdict derived from the selected criteria.

MIKE SHE-Hydro River had the highest wetland representation compatibility for the Kromme unchanneled valley bottom wetland, with a score of 12 points (

*Table 4*) because of realistic and spatially explicit options and processes (Table 3). There are several ways of implying wetland conditions in MIKE SHE-Hydro River within the floodplain with runoff and river inflows or as geographically isolated wetlands (GIWs) receiving runoff. The wetland configuration can be created in the model through the explicit elevation and topography inputs. For RWs, the modelling tool is able to characterise the case study wetland within the floodplain, receiving surface and subsurface runoff and upstream river inflows, and the vegetated land mass storage medium. The movement of water through the wetland, across the river floodplain, and due to a grid-based spatial scale, to the surrounding catchment cells representing the alluvial aquifer based on the hydraulic gradient is true to the physical wetland's context. Multiple inflow and outflow components and pathways, as well as storage properties and regulatory processes equipped the simulated wetland with options for modelling the palmiet wetland.

ACRU's wetlands characterised the wetland typology and storage media well allowing moderate compatibility to represent the physical wetland. The ACRU wetland options were similar. Both HRUs conceptualised the physical wetland as riparian, vegetated land masses with topographical dependence moderately implemented with the wetland as the most downstream component in the routing network. However, the riparian zone HRU outperformed the wetland HRU in terms of the flow pathways entering the wetland. The shortcoming of the wetland HRU model structure was that all wetland inflows of surface water and groundwater were received via the river channel and overbank spilling, or would be largely user-defined and consistent for the whole simulation period. The riparian zone HRU could split the wetland inflows into flows received as surface water in the river channel and as groundwater received in the wetland subsoil. This is a realistic representation of runoff contributions to the physical wetland.

Model documentation suggests that the parameter distributing how much upstream groundwater enters the wetland subsoil in the riparian zone HRU should be based on the wetland's extent in the floodplain area. The case study wetland was within, and spanned, the floodplain area, so this parameter was configured with the maximum value. Reducing the parameter could divert some upstream groundwater to channel and subsurface inflows.

*Table 3. Summary of wetland concepts in selected simulated wetlands compared to the case study wetland*

| Modelling tool | Physical wetland | ACRU | | MIKE SHE-Hydro River | | WRSM-Pitman | SWAT | |
|---|---|---|---|---|---|---|---|---|
| **Component** | | Wetland HRU | Riparian zone HRU | Overbank spilling | Flood zones | Comprehensive wetland module | Wetland HRU | Pothole HRU |
| **Basic premises** | Unchannelled valley-bottom wetland | A low-lying area that intercepts flow from the contributing upslope area and is adjacent to a river and receives channel overflow | Same as wetland HRU but receives upstream GW in subsoil (i.e. B-horizon) in proportion to its spatial extent in the riparian zone (user selected proportion) | Depressional land unit where water can collect, generally in close proximity to the channel and may be flooded by river overflow | Depressional land unit specified as a zone where inundation from river overflow can occur | In- or -off channel storage systems in a river network | Conceptual open water reservoir with spatially variable surface area, receives surface and subsurface* runoff from upslope areas | Depressional area where the river is poorly defined, and water collects as a function of low elevation and semi-impermeable substrate |
| **Dependence on topography** | Yes<br>Located in valleys, formation is geologically and geomorpho-logically influenced | Yes<br>Depressional, downstream component in the catchment configuration | Yes<br>Depressional, downstream component in the catchment configuration | Yes<br>Dependent on explicit elevation to create surface storage and area where river-wetland overflows/ exchanges can occur | Yes<br>Dependent on explicit elevation and floodplain zone delineation to create surface storage and area where river-wetland overflows/ exchanges can occur | Yes<br>Downstream component in the river network (channel provides runoff from contributing area to the wetland) | Yes<br>Fixed as upstream storage units within the subcatchment (portion of subcatchment runoff goes to wetland then into river) | Yes<br>Dependent on elevation to define depressional storage and route outflows |
| **Type of wetland** | Riparian | Riparian | Riparian | Riparian (User defined) | Riparian (User defined) | Riparian (User defined) | Geographically isolated | Geographically isolated |
| **Designed for** | | Channel-fed wetlands | GW- & SW-fed wetlands with veg accessing GW | Any riparian or meandering floodplain wetlands | Any riparian or floodplain wetlands | Unspecified, large wetlands | Unspecified | Playa lakes, artificially impounded fields or lakes |

* in this table, subsurface refers to runoff generated from the soil profile or GW, it may represent interflow, delayed surface runoff or GW outflows

| Modelling tool | Physical wetland | ACRU | | MIKE SHE-Hydro River | | WRSM-Pitman | SWAT | |
|---|---|---|---|---|---|---|---|---|
| Component | | Wetland HRU | Riparian zone HRU | Overbank spilling | Flood zones | Comprehensive wetland module | Wetland HRU | Pothole HRU |
| Water balance | **Inflows**<br>- Rainfall<br>- River and tributary inflows (SW and GW)<br>- Alluvial interflows<br><br>**Outflows**<br>- Storage outflows (visually observable surface runoff, may include variable composition of interflow and groundwater)<br>- Recharge to alluvial aquifer<br>- AET (from surface water, soil, shallow GW /saturated zone)<br>- Interception | **Inflows**<br>- Rainfall<br>- Channel flow (composed of upstream SW and GW)<br><br>**Outflows**<br>- Surface runoff<br>- Interflow (delayed surface runoff)<br>- Groundwater outflows<br>- AET (soil water evaporation, transpiration, not from shallow GW)<br>- Interception | **Inflows**<br>- Rainfall<br>- Channel flow (composed of upstream SW)<br>- GW inflows into the subsoil<br><br>**Outflows**<br>- Surface runoff<br>- Interflow (delayed surface runoff)<br>- Groundwater outflows<br>- AET (soil water evaporation, transpiration, not from shallow GW)<br>- Interception | **Inflows**<br>- Rainfall<br>- Upslope surface runoff<br>- Channel/river overflow<br>- Capillary rise<br>- GW & interflow inflow from saturated zones<br>**Outflows**<br>- spilling to downslope land or river cell<br>- Recharge to aquifer/ saturated zone<br>- AET (from surface water, soil, shallow GW /saturated zone)<br>- Interception | **Inflows**<br>- Rainfall<br>- Upslope surface runoff<br>- Channel/river overflow<br>- Capillary rise<br>- GW & interflow inflow from saturated zones<br>**Outflows**<br>- Spilling to downslope land or river cell<br>- Recharge to groundwater reservoir<br>- AET (from surface water, soil, shallow GW /saturated zone)<br>- Interception | **Inflows**<br>- Rainfall<br>- River inflows (containing SW and GW from upstream modules)<br><br>**Outflows**<br>- Return flows to the river<br>- AET (open water algorithm) | **Inflows**<br>- Rainfall<br>- Surface and subsurface runoff from surrounding HRUs<br><br>**Outflows**<br>- Surface runoff to river<br>- Seepage flowing back to the river<br>- AET (open water rates) | **Inflows**<br>- Rainfall<br>- Surface and subsurface runoff from surrounding HRUs<br>- Irrigation<br><br>**Outflows**<br>- Surface runoff to pothole<br>- Seepage to underlying soil profile<br>- AET (open water algorithm adjusted by vegetation shading) |
| Storage media | Vegetated land mass (e.g. saturated soils) | Vegetated land mass with soil profile<br>May include GW storage, no surface storage | Vegetated land mass with soil profile<br>May include GW storage, no surface storage | Vegetated land mass with soil profile | Vegetated land mass with soil profile | Open water body that evaporates at rates similar to vegetation, Acts as a unified SW unit | Open water body that evaporates at rates similar to open water, Acts as a unified SW unit | Open water body on top of a land mass that evaporates at rates similar to vegetation, Acts as a unified SW unit |

| Modelling tool | Physical wetland | ACRU | | MIKE SHE-Hydro River | | WRSM-Pitman | SWAT | |
|---|---|---|---|---|---|---|---|---|
| **Component** | | Wetland HRU | Riparian zone HRU | Overbank spilling | Flood zones | Comprehensive wetland module | Wetland HRU | Pothole HRU |
| **Storage regulation** | Consistent subsurface inflows from regional TMG and local alluvial aquifers and side tributaries,<br>SW river inflows which include GW, diffuse flows through the wetland,<br>large wetland storage and water retention supporting consistent outflows from the wetland to the river and possible recharge to the alluvial aquifer,<br>dense vegetation coverage,<br>clayey topsoil and deep soil with large soil water storage,<br>all river inflows pass through/enter the wetland | All SW and GW from upstream subcatchment routed to the wetland as river inflows,<br>Channel capacity threshold determines flows above which spillage occurs onto the wetland,<br>Topsoil moisture content dictates the generation surface runoff or infiltration of rain & channel spill,<br>GW storage releases water in proportion to current volume at a user-specified rate | Same as wetland HRU except for upstream GW inflows received in the subsoil | Infiltration rate controls the rainfall infiltration or detention storage generation,<br>Water moves from upslope cell to adjacent or downslope cell through a varied topography,<br>Regulated by elevation and water level gradients between cells | Infiltration rate controls the rainfall infiltration or detention storage generation,<br>Water moves from upslope to downslope cross-section through a relatively uniform topography per cross-section,<br>Regulated by a water level gradient between sections in the floodplain | River inflows into wetland are activated by channel capacity threshold that allows spillage onto the wetland, with a proportion passing through the wetland or entering the wetland storage,<br>Current wetland storage relative to maximum storage determines whether storage is retained in the next time step or released at a rate in proportion to excess volume | Inflows regulated by amount of runoff produced by other HRUs in the subcatchment and as a user-defined proportion of subcatchment area feeding the wetland,<br>Storage regulated by maximum wetland storage and saturated hydraulic conductivity | Inflows regulated by amount of runoff produced by other HRUs in the subcatchment and irrigation requirements,<br>Storage regulated by underlying soil water content and maximum storage capacity that will activate seepage or overflow |

*Table 4. Compatibility verdict reflecting wetland representation in the wetland models for the case study wetland*

| **Modelling tool** | **ACRU** | | **MIKE SHE-Hydro River** | | **WRSM-Pitman** | **SWAT** | |
|---|---|---|---|---|---|---|---|
| **Component** | **Wetland HRU** | **Riparian zone HRU** | **Overbank spilling** | **Flood zones** | **Comprehensive wetland module** | **Wetland HRU** | **Pothole HRU** |
| **Basic premises** | 2 | 1 | 2 | 2 | 2 | 0 | 1 |
| **Dependence on topography** | 2 | 2 | 2 | 2 | 2 | 1 | 1 |
| **Type of wetland** | 2 | 2 | 2 | 2 | 2 | 1 | 1 |
| **Water balance** | 1 | 1 | 2 | 2 | 1 | 1 | 1 |
| **Storage media** | 2 | 2 | 2 | 2 | 1 | 0 | 1 |
| **Storage regulation** | 1 | 1 | 2 | 2 | 1 | 1 | 1 |
| **Total** | 10 | 9 | 12 | 12 | 9 | 4 | 6 |
| **Suitability rank** | 2nd | 3rd | 1st | 1st | 3rd | 5th | 4th |

* darker shading indicates higher score and compatibility of the wetland when model configured for the case study wetland

** all scores are related to the features and properties of wetland models described in Table 3 and section 3.3.1.

## 3.4. Discussion

Simulated wetlands in hydrological models vary in structure but share a basic common definition centred on storage regulation (i.e. a series of temporally varying inflows and outflows). Prioritising simulated wetlands in the model selection process is complicated by the large number of modelling tools available, each with their own mathematical solution and representation of the wetland, and different wetland-catchment interactions. These factors also contribute to the omission of a simulated wetland’s compatibility for the physical wetland. The credibility of a simulated wetland is rarely attributed to the realism maintained in the model, but rather to the catchment-scale model’s ability to replicate observed data.

The analyses addressed wetland representation as a function of the wetland characteristics and processes in seven simulated wetlands’ relative to the case study wetland. A framework for comparing model wetlands from several modelling tools was tested on an unchannelled valley-bottom wetland. The results identified wetland routines and setups available in modelling tools and assessed simulated wetlands in terms of differentiating wetland features which are common in all wetland models.

The assessment suggests that all simulated wetlands exist within a hydraulic gradient relative to the surrounding catchment or river and operate based on the current wetland storage volume, inflow rates and outflow rates to represent the wetland's relationship with its surrounding catchment and river. A key finding from the results was that simulated wetlands have differing sets of fixed (e.g. the wetland storage media and algorithms) and flexible properties (e.g. inflow pathways and proportions retained in the wetland or passing through the wetland and conditions for outflows from the wetland). Modelling tools with multiple options for constructing a simulated wetland are more likely to represent the processes of a physical wetland. However, most modelling tools have a specific wetland type and processes embedded into their model structure which limits the tool’s compatibility to specific wetlands and conditions. In such cases, wetland parameters are marginally helpful in implying alternative wetlands and processes.

### 3.4.1. Compatibility between the physical and simulated wetland

Considering the model type, more physically-based wetland models (e.g. MS-MHR and ACRU) outperformed wetland representation of the Kromme wetland compared to the conceptual wetland models (e.g. SWAT and WRSM Pitman). In the context of the wetland model's scope for wetland types, the wetland model specificity within a modelling tool resulted in either few or many similarities between the case study wetland and wetland model. In the conceptual wetland models with few similarities, there were many missing processes and features relative to the case study wetland. In the physically-based models with many similarities between the model and case study wetland, it was possible to create a wetland model with wetland conditions and processes that are similar to the Kromme wetland. To illustrate this, MS-HR is a physically based, multi-optional model allowing the tool to be setup for various wetland types and potential flows. The high level of detail and spatially explicit processes in MS-HR allowed all of the physical wetland's inflows and outflows to be incorporated in the wetland models. In addition to this, the wetland domain in MS-HR used the same detailed process representations as the catchment cells. Less compatible simulated wetlands were identified in the conceptual modelling tools.

The conceptual modelling tools focused on implementing a specific wetland type. In terms of this case study wetland, ACRU's model structure and parameters were designed for riparian wetlands and yielded moderately compatible wetland representations although the model is conceptual. In addition to using the same process representations as the catchment HRUs, the wetland was specialised to include flow pathways associated with wetlands. These pathways included overbank spilling and access to upstream groundwater in via subsurface pathways. On the other hand, SWATs wetland models (i.e. the wetland HRU) prioritised conceptualising downstream ponds or lakes and marshes that were partially disconnected from the main channel reducing their compatibility for the Kromme wetland, especially for with the pothole HRU. Moreover, the SWAT wetland units did not inherit the same model complexity as the catchment and simply regulated the wetland storage by volume. In this case, this rigidity for a specific wetland type, storage and processes made the SWAT wetland models incapable of accurately conceptualising the Kromme wetland. Similar findings of improved wetland representation were found in Liu *et al*. (2016) when modelling the Prairie Pothole Region in the physically-based HydroGeoSphere with integrated surface water and groundwater flows. The advantage of multi-optional model structures was highlighted in a recent review investigating strategies to improve process and scale representations in hydrological models where Sidle (2021) identified flexible model structures as a precursor for improving flow pathways in a model. Fixed wetland model structures focusing on wetlands different from unchannelled valley-bottom wetland limited wetland representation.

Wetland representation improves when fixed properties from the model are compatible with the physical wetland. The wetland typology, storage regulation and flow pathways may be slightly modifiable with setup choices (e.g. the wetland's location in the catchment which implies the area contributing to the wetland inflows and where the wetland discharges to) and parameter choices (e.g. wetland storage capacities, infiltration properties, inflow and outflow rates) from the modeller. However, the simulated wetland's storage media and water balance components are fixed properties from the model design.

Excellent to acceptable representation scores, in reference to the case study wetland, were assigned to models with compatible typologies and water balance components in combination with the presence of a soil profile in the wetland storage conceptualised in MS-HR and ACRU. Moderate wetland representation was found in the comprehensive wetland for the WRSM-Pitman with a compatible typology and water balance components despite an incompatible wetland storage media. Poor wetland representation was associated with the joint conceptualisation of the wetland storage as a water body, missing water balance components and incompatible routing of wetland outflows (e.g. outflows remained in the pothole HRU and depend on the soil profile which is external to wetland's water balance) despite implying riparian wetland typology (e.g. implying the wetland a downslope wetland with all upstream HRUs runoff entering SWAT's wetland HRU or all inflows from user-selected HRUs in SWAT's pothole HRU).

The wetland storage media represented in WRSM-Pitman's comprehensive wetland and SWAT's wetland models could not be modified with setup or parameter choices to represent the land mass of the case study wetland. In terms of runoff outflows, the conceptualisation of the wetland storages as a unified surface water body did not allow for the differentiation of surface water and groundwater outflows, nor the specification of subsurface properties in the soil or aquifer that may influence the water retained or released from the wetland. The latter could only be implied in an outflow rate parameter in WRSM-Pitman and seepage regulated by the saturated hydraulic conductivity in both of SWAT's simulated wetlands and soil water content in the underlying soil profile in the pothole HRU. This decreased their compatibility scores. The results suggest that the wetland storage media and the potential inflows and outflows are fixed in simulated wetlands. The compatibility of fixed wetland features significantly impacts the overall suitability and realism of the simulated wetland.

Many studies refer to the importance of the wetland storage and flow pathways between the wetland and catchment. Examples of such studies exist for model development (Rahman *et al*., 2016; Evenson *et al*., 2018; Dash *et al*., 2020), establishing model performance (Evenson *et al*., 2016; DeVries *et al*., 2017) and understanding model behaviour (Gray, 2011; Muhammad *et al*., 2019). The fundamental definition of simulated wetlands as temporally variable storage units alludes to the storage regulation's dependence on the volume of water stored in the wetland at any time step. This suggests that wetland storage plays a big role in determining whether streamflow attenuation and augmentation is simulated and the hydraulic gradient with the adjoining rivers and surrounding catchment. The same reasoning (i.e. the importance of the wetland volume in streamflow regulation) is true of physical wetlands where the extent of streamflow attenuation or supplementation depends on the antecedent volume of water in the wetland and the subsequent wetland storage deficit (Morris and Camino, 2011).

In terms of the runoff in the wetland water balance, runoff can be differentiated into surface and subsurface runoff for simulated and physical wetlands. The case study wetland is known to receive and release both surface and subsurface outflows (Tanner *et al*., 2019; Smith, 2019; Ollis *et al*., 2013). The WRSM-Pitman and SWAT wetland models which simulated the wetland storage as a single water body were unable to distinguish runoff outflows into surface and subsurface outflows. Alternatively, the differentiation of the wetland profile in ACRU and MS-HR could simulate the variable composition of runoff inflow and outflows, maintaining a higher degree of hydrological realism. This highlights how the wetland storage and water balance components, which are relatively fixed in the simulated wetland, can significantly affect wetland representation.

Conceptual models achieve wetland representation when applied to wetland types incorporated in the model design. The evidence from this analysis and existing literature appear to suggest that physical models are better at achieving wetland representation. However, wetland representation still depends on the compatibility of the simulated wetland with the cases study wetland and can be achieved in conceptual models. For example, there is evidence for the SWAT models, which were found unsuitable for the unchannelled valley-bottom wetland, to be exceptionably suitable for geographically isolated, pothole and perched wetlands (Muhammad *et al*., 2019; Lee *et al*., 2018b; Yeo *et al*., 2019a; Lee *et al*., 2019). On the other hand, ACRU is a conceptual model which prioritises riparian wetlands in the model design of the riparian zone and wetland HRUs. Thus, demonstrating that wetland representation can be achieved in conceptual models with relevant parameters and when the wetland storage and typology are compatible. This also reiterates the idea of wetland representation's correlation with fitting the physical wetland into an existing, inflexible model or the ability to create the physical wetland in a multi-optional tool or compatible model.

A previous review and application of the WRSM-Pitman wetland to a floodplain wetland found that wetland representation was insufficient due to no seep-groundwater processes and the spill-and-fill, reservoir concept being suitable for large-scale wetlands (Maherry *et al*., 2017). This is different from the moderate compatibility found for the comprehensive wetland when applied to the case study wetland where there was no seepage due to the underlying geology. The impact of the case study wetland's spatial scale on model performance is yet to be determined via quantitative modelling. From the qualitative comparison, the spatial scale of the wetland and catchment in WRSM-Pitman did not prohibit conceptualising the case study wetland using the comprehensive wetland module.

The AET *al*gorithm is another critical, fixed wetland feature in all simulated wetlands and open water AET was not a suitable algorithm for the unchannelled valley-bottom wetland. The case study wetland is associated with AET firstly, because of its large subsurface storage which is consistently sustained by groundwater inputs from the regional and alluvial aquifer; and secondly, because of vegetation composition (i.e. the dense palmiet and invasive wattle). The high water use associated with wetland type needs to be captured in the simulated wetland. AET can be calculated for surfaces representative of open water or vegetated surfaces. In addition to this, different surfaces have different evaporation rates with open water generally yielding larger losses (Allen *et al*., 1998; Savage *et al*., 2017) and different calculation methods produce different estimates of AET (see Section 3.3.1. and 3.3.2.) In terms of peatland wetlands, AET fluctuations have been linked to the vegetation composition and land management (Ahmad *et al*., 2020) as well as the groundwater table and interannual rainfall variability (Ahmad *et al*., 2021; Schwarzel *et al*., 2006). In the wetland models analysed, the A*Et al*gorithm was fixed. The simulated wetlands which implemented for water losses with an algorithm for vegetation and soil water availability for evaporation had higher compatibilities than simulated wetlands with open water algorithms. The different calculation methods are likely to introduce variable simulations of water use from the wetlands. Simulated wetlands are built on the water balance as the fundamental principle. Inaccurate AET outflows could impact the predicted wetland storage volume and the storage deficit which subsequently affects the interception of flood waters (i.e. streamflow attenuation) or the onset of wetland outflows (i.e. streamflow supplementation). Unchannelled valley-bottom wetlands require models which can simulate AET from losses from the vegetation and variable access to subsurface water storage (viz. the changing water availability for AET in response to vegetation density and composition, climate and groundwater conditions).

### 3.4.2. Potential implications of wetland representation on model performance

Model complexity does not guarantee model performance to the extent that hydrological realism does. Several large-scale studies have investigated the impact of spatial scale on model performance using distributed and lumped models, the impact of model type using conceptual and physical models on model performance of streamflow and surface-groundwater representations (Breuer *et al*., 2009; Smith *et al*., 2004; Smith *et al*., 2012; Carpenter *et al*., 2006; Reed *et al*., 2004; Garavaglia *et al*., 2017; Kollet *et al*., 2017; de Boer-Euser *et al*., 2017; Vansteenkiste *et al*., 2014; Darbandsari and Coulibaly, 2020; Baroni et al., 2019). According to Muhammad *et al*. (2019), the spatial scale of the modelled wetland appears to not be as much of an issue as the match between the processes described in the model and the input data. Although the wetland representation was high in the physically-based modelling tools, Devia *et al*. (2015) noted that the low practicality of such models from high data, computational and setup requirements limit their usage. Regarding the model type, research based on case studies (i.e. modelling examples in various contexts) showed that there are three perspectives on the issue. Some research observed that one model type leads to better results (Jaiswal *et al*., 2020; Liu *et al*., 2016). Other research found that neither model type improves model performance (Devia *et al*., 2015) nor does model complexity or simplicity equate to better model performance (Tegegne *et al*., 2017). In terms of modelling case study wetlands, previous research on wetland-river exhanges and modelling wetlands hydrology and hydraulics suggest that high degrees of hydrological realism be incorporated in the modelled wetland's characteristics and processes (Makungu and Hughes. 2021); and guidance on parameter choices (Hughes *et al*., 2013). This suggests that conceptual and physical models have the potential to produce reliable model performance on the condition that the simulated environment accurately describes the physical wetland. Worst case scenario is that the models could provide the correct output for the wrong reasons or still predict the incorrect streamflow despite best efforts to have hydrologically realistic processes. However, focusing on the wetland models' potential and current representation, the wetland models from physical (i.e. MS-MHR) and conceptual tools (e.g. ACRU and WRSM-Pitman) with high hydrological realism for the Kromme wetland are more likely to have acceptable model performance. Expectations about model performance require evidence from simulating the catchment.

### 3.4.3. Effectiveness, reusability and efficiency of comparative method

Although this qualitative assessment of model structure is insufficient to determine if and when a modelled wetland would predict streamflow attenuation or augmentation, the selected criteria sufficiently describe wetland features related to streamflow regulation. The results, with some measure of confidence, suggest that the wetland compatibility scores are likely to lead to better model performance for the unchannelled valley-bottom wetland. However, a meta-analysis of factors affecting the extent of streamflow regulation in several studies reported that wetland characteristics alone did not correlate or predict the extent of streamflow regulation (Findlay and Kadykalo, 2016). Therefore, wetland characteristics alone don't dictate the streamflow regulation impact a wetland would predict. Rather, the wetland's properties together with the surrounding catchment, climate and specific pattern of weather events in the time period being assessed affect whether certain streamflow regulation impacts occur. Further conclusions about whether a model predicts attenuation or supplementation requires setting up the wetland model in a catchment and with a climate timeseries.

The criteria for evaluating wetland models sufficiently described the wetland unit in one or more tools. All seven wetland models could be defined in terms of the criteria (i.e. an underlying concept, location in the catchment, wetland typology, wetland water balance and storage regulation processes). This demonstrates that the criteria were not only standardised but applicable for several modelling tools and model types (i.e. physical or conceptual) of interest. Differences in wetland features relative to the physical wetland, whether from setup or parameter options and fixed or flexible wetland properties, were also observable. Concerning tool-to-tool comparisons, each models strengths and weaknesses were noticeable. The description of simulated wetlands relied on a collection of resources (including expert opinion, technical reports, several publications and theses) in addition to the anticipated use of the tool's documentation to consolidate the wetland models. Overall, the method effectively outlined the capabilities of simulated wetlands in each modelling tool and the wetland's context in the catchment without the need for fieldwork or modelling.

### 3.4.4. Limitations

The current scope of the analysis did not consider the other six HGM wetland types. Wetland representation was assessed for one wetland type for several models and needs to be validated for other wetland types and model configurations. The small spatial scale (catchment < 50 $km^2$ and wetland area < 1.5 $km^2$) was another limitation of this study. Water management assessments usually occur at the quaternary scale which is notably larger than the case study catchment. It is not certain if the same wetland characteristics and processes are applicable for larger scales (viz. if wetland representation can be fairly assessed and compared on larger wetlands or which wetland features become more or less important for wetland representation at larger spatial scales). More importantly, wetland representation was not considered in terms of the wetland characteristics and processes interacting to produce streamflow attenuation and supplementation.

The framework for qualitatively comparing wetland models could be limited by the scoring system and wetland model input adopted in the method. The scoring system could use an increasing points approach (i.e. start the compatibility score at zero and increase the score with each compatibility detected) or decreasing points approach (i.e. start the compatibility score at an arbitrary maximum value and decrease the score with incompatibilities detected). The current methodology used the decreasing points approach which favours the detection of incompatibilities but having a limited maximum value of compatibility allows the scores to remain comparable between tools, as opposed to infinitely increasing in the increasing points approach. Although incompatibilities are necessary to identify to ensure the model has suitable properties and processes for the physical wetland, it is possible that the scoring system is too critical and sensitive to incompatibilities. In terms of completing the wetland model comparison by different users, experts were reported to give different scores of wetland ecosystem services assessed qualitatively (Walters *et al*., 2021). If the qualitative approach adopted for comparing wetland models was completed by different users, it is possible that different users could yield different compatibility scores. The possibility for different compatibility scores and the magnitude of variability is not yet defined. Furthermore, the applicability of the current criteria is yet to be tested for other types of wetlands which may have other essential features, characteristic and processes (Rebelo *et al*., 2019a).

Regarding the wetland models assessed, the model structure and parameters options were considered. The model structure is inherent in the model design and relatively objective. Parameter values are subjective to the modellers choices and have a range of possible values. Extreme values were selected for parameters in the wetland models related to the wetland typology and location. This is related to parameter values in the models. For example, in the riparian zone wetland HRU, 100 5 of upstream groundwater inflows were routed to the wetland subsoil. In reality, this value could be adjusted to any user defined value. Furthermore, when modelling, it is possible that different parameter values could be appropriate for the wetland setup. In this case, changing the parameter values could potentially produce different compatibility scores and a wetland model which is more similar or dissimilar to the unchannelled valley-bottom wetland.

### 3.4.5. Recommendations for implementation and future research

Implementing the comparison of wetland models could facilitate the model selection process in catchment water supply assessments. Practitioners involved in catchment-scale modelling for water supply management have limited time, resources and modelling preferences. These restrictions sometimes lead to the omission of wetlands in modelling, parameter choices that do not reflect the wetland realistically (i.e. over parameterisation) (Maherry *et al*, 2017), or the selection of legacy models irrespective of the wetland's suitability (Kundzewicz *et al*., 2019). The method used in this study promotes the inclusion of wetlands in catchment models by focusing on understanding the wetland unit and its context in the catchment on a case-by-case basis. Identifying the key wetland features also develops a model concept that can be applied to the modelling process and facilitates the interpretation of the modelled output.

The assessment can be looked at as a preliminary step to modelling the wetland and reduces the time requirements of modelling. The standard criteria and repeatable nature of the method enables the method to build a knowledge base of wetland models in different tools and for different wetland types. This is an additional time saver for other modellers in academic or practical settings.

Increasing the scope of the assessment could advance the qualitative analysis. Moving forward, the method used in this study needs to be applied to other wetland types and modelling tools in different climates and geomorphological settings. The sensitivity of the compatibility scores to different users could be investigated as part of future research objectives. Applying the comparison to other wetland types may introduce additional criteria or highlight other wetland features which are critical to the model's suitability. Furthermore, different scoring approaches could also be tested to minimise potential subjectivity in the method. Most importantly, the findings on wetland representation in this assessment need to be paired with modelling the wetland in its catchment and identifying the wetland's impact on streamflow regulation.

## 3.5. Concluding summary

Simulated wetlands have defining features that can be used to navigate wetland representation in different modelling tools and models. The distinguishing features include the modelling tool's overarching concept of a wetland, the location of the wetland in the catchment, the wetland typology relative to the floodplain, the wetland storage media, the wetland's water balance and processes regulating the wetland storage.

Hydrological realism for an unchannelled valley-bottom wetland underlain by impervious substrates was represented well in physical and conceptual models. Additionally, wetland representation improves in multi-optional tools or when the fixed model structure properties are compatible with the physical wetland. In this case, fixed model properties identified were the wetland storage media and wetland typology. Furthermore, wetland AET routines that did not simulate soil water evaporation and transpiration from water accessed in the soil moisture reserves and groundwater reduced the compatibility scores of the wetland models.

The analysis conducted has potential to facilitate the model selection process, preliminary modelling and the inclusion and prioritisation of wetlands in hydrological modelling. The comparison allows wetland models to be selected based on their ability to conceptualise the correct and exiting processes in a physical wetland (i.e. correct simulations for the correct reasons). This could improve the accuracy of modelled water supplies and the implications of various catchment changes on wetlands and water yields (e.g. land use change, climate change, restoration efficacy and degradation impacts). Real-time simulations from modelling the wetland are necessary to identify how the wetland characteristics and processes interact to simulate streamflow regulation (viz. the wetland’s hydrological function).

# Chapter 4: Modelling the wetlands

## 4.1. Introduction

Chapter 3 drew two key learning outcomes from the comparison between physical and simulated wetland concepts: simulated wetlands have different underlying concepts, and these concepts may not be suitable for a particular physical wetland. The next step is to determine how the modelled wetlands behave within the catchment and climate context. An assertion that one model is better than another invites contention and debate, both globally and locally. In this study, for each model, wetland representation is credited to the model's ability to capture the physical properties, processes and functions of the real wetland.

The purpose of this chapter is to assess wetland representation of the upper Kromme wetland based on simulated outputs. The modelling completed in this chapter addresses the limitations in the qualitative analysis which did not consider wetland representation in terms of the wetland's influence on streamflow given its specific catchment and climatic context. Furthermore, the regulatory role of simulated wetlands can only be identified by modelling the catchment.

### 4.1.1. Model performance

Traditionally, model performance is based on a model's ability to replicate measured streamflow data. Streamflow is a measure of the catchment water availability after all inflows and outflows have occurred in the catchment (Juniati *et al*., 2018; van Gaelen *et al*., 2017). As a result, most modelling studies, including comparative modelling, calibrate and validate models using observed streamflow records (Krysanova *et al*., 2017; Vansteenkiste *et al*., 2014). In broader perspective and considering model design, water balances are the fundamental principle of hydrological models (Ruqayah, 2018; Singh, 2018) and ideally would be part of model performance assessments. Although streamflow is readily available compared to other components in the water balance, according to Beven (2019), model performance is often limited by the quality of the observation records and the inherent errors in the model's simplification of reality.

Despite data availability and record length limitations, there are benefits to including other variables in the water balance to support the model credibility conclusions from streamflow-based model performance assessments. Recent modelling studies are showing a shift towards verifying models are considering with streamflow and other variables in the water balance. The main benefit of including other variables in model performance assessments is the reduced risk of accepting a model which may simulate other sections of the hydrograph or water balance incorrectly. Streamflow performance can be variable. For example, a multi-model (i.e. using four models), multi-site (i.e. 1 000 catchments) comparative modelling study by Lane *et al* (2019) showed that models can perform well for different flow segments in the hydrograph, different catchment characteristics and different components in the water balance.

Furthermore, good model performance for streamflow does not guarantee sound model performance for the whole water balance and good model performance in one catchment does not ensure that the tool perform well in other catchments. An additional benefit of rigorous model performance based on several water balance components reveals whether the water balance is accurately modelled. With reference to literature, an earlier review about model usability in a range of catchments and environments by Gupta *et al*. (2014) termed the ability of a model to replicate various components of the hydrological cycle as hydrological realism. Consequentially, multi-variable model performance assessments ensure that credible models and output facilitate the transfer of reliable information from the model to the processes and decisions the model informs. In terms of wetland modelling, this shift towards more in depth assessment of hydrological realism was embraced as early as 2004 where Acreman *et al*. (2004) recommended identifying and quantifying processes that move water into and out of the wetland (referring to the collective water movement as transfer mechanisms).

Considering what does work for model performance assessments, assessing a catchment and wetland water balance variable is a proven benchmark for establishing a models credibility. In the same way streamflow is the typically the variable of focus from the catchment water balance, wetland storage can be used as the focal variable of the wetland water balance and a suitable variable for qualifying the wetland's model performance. Evenson *et al*. (2018) reported that evaluating the model performance based on a target variable from the catchment water balance qualified too many models as acceptable. Adding a variable from the wetland water balance, in this case wetland storage, narrowed down the selection of plausible models and confirmed hydrological consistency for the catchment and wetland simulations. However, wetland storage is rarely monitored. Considering other avenues to source this data, DeVries *et al*. (2017) presented a method to estimate remotely sensed wetland storage but reported continuing difficulties to measure subsurface water volumes in wetlands that are vegetated and have significant water storage in soil media. At this stage, accurate estimates of remotely-sensed wetland storage are limited to wetlands with mostly open water surfaces, making it unsuitable for the case study wetland. Other variables, such as AET, could be considered for performance assessment if observational datasets exist.

These studies highlight the possibility of an inaccurate wetland water balance in a model with an acceptable catchment water balance and the subsequent necessity to estimate both water balances.

### 4.1.2. Study design for comparing catchment-wetland models

Concerning wetland model comparisons, some studies focus on the model setup to estimate the influence of the wetland on the simulated catchment streamflow. One way of doing this is with the use of increasing spatial detail in the model setup. Muhammad *et al*. (2019) investigated the effect of increasing the spatial representation of wetlands in the Canadian Prairie region in a lumped, semi-distributed and fully-discretised model. The study found that model performance for catchment streamflow was best in the fully-discretised model. Another way of adjusting the model setup to account for wetlands is by changing the catchment drainage of land uses into or out of the wetland, and the water flow pathways in the catchment. For example, some studies have used catchment configuration and routing adjustments to improve the representation for a specific type of wetland (Rahman *et al*., 2016; Evenson *et al*., 2016; Liu *et al*., 2008).

Comparing model output representing different scenarios is another approach used for detecting wetland influences on catchment streamflow. Examples include wetland loss and restoration (Jones *et al*., 2017), using one wetland type per simulation (Lee *et al*., 2018b) and varying the wetland's location in the landscape (i.e. riparian or non-riparian) (Fossey *et al*., 2016). Furthermore, Kadykalo and Findlay (2016) reviewed the wetland studies focusing on flow regulation services and highlighted three common scenario designs in modelling studies and focal points from monitoring: the first design compared scenarios before and after an impact; the second case was the comparison of flows from a scenario with an unmodified versus modified wetland; the third case compared more than one wetland exposed to an impact and compared this to a wetland that is relatively pristine.

In terms of quantifying the wetland's influence on the catchment hydrology or streamflow, the metrics for flow regulation services provided by a wetland can be associated with the model setup or customised to the study's context. For wetland modelling studies and metrics dependent on the model setup, paired scenarios comparing the model output from a scenario with and without a wetland are a popular choice (Wu *et al*., 2019; Fossey and Rousseau, 2016; Yeo *et al*., 2019a). The results from these scenarios indicate when the wetland component attenuated or supplemented streamflow. Alternatively, metrics are specific to flows. For wetland modelling studies focusing on selected flows irrespective of the study design, Kadykalo and Findlay (2016) found that flow regulation is often reported as the reduction in flooding and total runoff volumes or the increase in low flows. General modelling studies have embraced hydrological realism with hydrological signatures and the selection of several model performance statistics covering all flow segments of the hydrograph (i.e. high, moderate and low flows) (Sawicz *et al*., 2011; Pool *et al*., 2017). Lastly, metrics can be specific for the research objective. For example, a custom metric for flood attenuation was in Rebelo *et al*. (2019) where streamflow responses to high rainfall events were related to the proportion of wetland vegetation in the valley. Another custom metric of wetland attenuation was presented in terms of outflow rates with regards to inflows, and the area-volume and storage-inflow relationship for a wetland (Hughes *et al*., 2013; Makungu and Hughes, 2021).

In hydrological modelling under changing climates, drought and flood events are rarely assessed in wetland modelling studies although a wetland's influence is expected to be emphasised during these times. Streamflow attenuation and supplementation caused by wetlands has been linked to extreme hydroclimatic events, in that some wetlands have been found to reduce flood flows and increase water supply during droughts. However, the current literature and research in wetland modelling studies only consider the climate by modelling multiple wetlands in different geographical regions (Mandlazi, 2017) or comparing extreme flows from simulations using historical and projected climates (Fossey and Rousseau, 2016). This demonstrates that deductions made from modelling in past or future climates are made without considering the hydrological response or influence of the wetland model on catchment streamflow for historical events.

Furthermore, most studies consider climate variability with time-specific analyses and seasonality. In terms of monitoring and modelling groundwater dynamics, investigating streamflow responses in extremely a wet year has been conducted for a riparian-hillslope interface (Scheliga *et al*., 2018). Another case study monitored and modelled a peatland water balance during a drought (Streich, 2019). Regarding temporal variability and model behaviour under certain rainfall conditions, interannual climate differences were sometimes accounted for by analysing the streamflow of dry and wet years (Wu *et al*., 2019; Makungu and Hughes, 2021) or the long-term seasonality within the hydrological year (Mandlazi, 2017; Maherry *et al*., 2017). These modelling studies demonstrate that weather variability can be accounted for with event-specific analyses.

Ideally, a model of a wetland would be able to reliably predict the wetland's impacts on extremes in addition to long-term average impacts. Concerning the wetlands impacts during extreme flows, previous modelling and field studies have demonstrated that individual or networks of wetlands can attenuate floods and supplement low flows (Blanchette *et al*., 2019; Blanchette *et al*., 2022). In terms of model performance, some authors have highlighted the changes in model performance when the simulation deviates from the calibration period conditions (Bai *et al*., 2022) or average conditions which the model is configured for (Dai *et al*., 2010). Respectively, in the first case, concerns are raised from a calibration period which may not include the extremes where a wetland's impact on the catchment streamflow becomes prevalent; and, in the second case, an acceptable model fit with respect to average conditions does not ensure representation during extreme climate conditions. Regarding model configuration, Melsen *et al*. (2019) investigated the influence of a modeller's decisions concerning the model configuration and calibration and demonstrated the degree to which model output is sensitive to these choices. In particular, flood characteristics (i.e. maximum discharge, peak volume and timing) are most sensitive to the performance metrics selected for calibration, while drought characteristics (i.e. minimum discharge, the onset of the drought, deficit flow relative to baseflows for the whole period and the duration of the drought) are most sensitive to the flow conditions in the calibration period (viz. predominantly high or low flows). In terms of wetland modelling comparison studies, there is yet to be a study which focuses on model performance and behaviour during extreme hydrological events. Evaluating event-specific streamflow from wetland models would be a worthwhile endeavour considering the model output sensitivities to extreme climates and streamflow regulation occurring during, and in response, to extreme climatic events.

### 4.1.3. Current state of wetland modelling in South African catchments

Moving from the options for comparing wetland models and streamflow regulation simulations to the body of literature on modelling studies, previous wetland modelling studies in South Africa have focused on impact assessments or the goals of the research. Examples of such studies include Rebelo *et al*. (2015) which investigated the impacts of land-use and -cover changes on streamflow to advocate for wetland restoration. A later study focused on the ecological services provided by palmiet wetlands to motivate for the payment of services used (Rebelo *et al*., 2019a). In addition to these modelling studies, Tanner *et al*. (2019) conducted exploratory research on the hydrological and geomorphological function of palmiet wetlands involving site monitoring, hydrological modelling and water quality tracing and modelling to identify water flow pathways.

In terms of comparative wetland modelling, there are two modelling studies focusing on wetlands. To make a case for the inclusion of wetlands in catchment-scale modelling, Maherry *et al*. (2017) reviewed the simulated wetland concepts in five modelling tools, modelled two catchments in one modelling tool (SPATSIM-Pitman), reconstructed a rainfall-ET water balance in one catchment with the modelled output and presented twenty-one wetland conceptual water balance models for the seven HGM wetland types in Ollis *et al*. (2013). In the second study making a case for modelling wetland processes observed from monitoring, Mandlazi (2017) investigated catchment-wetland interactions by modelling four wetlands (typed as unchannelled valley-bottom wetlands and floodplains) in four different catchments using two modelling tools (namely, SPATSIM-Pitman and ACRU).

In terms of model performance standards, the aforementioned studies on South African wetlands demonstrate a willingness to include auxiliary variables of the water balance where data is available in model performance assessments. Investigating different variables in the water balance exceeds the general requirement to verify catchment streamflow and aligns with the emerging trend to assess the catchment water balance from hydrological models. However, both wetland-focused modelling studies defined model performance in terms of streamflow replication. In this way, the replication of supporting variables in the water balance did not have any bearing on whether the model was accepted or rejected. Rather, investigating other variables was used to confirm how realistic the models were. For example, hydrological realism was considered with the water balance reconstruction using remotely-sensed ET and model output (Maherry *et al*., 2017). In another study, Mandlazi (2017) expressed an interest in modelling and measuring groundwater flows. Furthermore, Tanner *et al*. (2019) partially validated the hydrological models using AET information and showed significant effort in estimating groundwater dynamics.

With reference to the evolving best practices for wetland model comparisons and model performance assessments, the effectiveness of model comparisons in the local studies was reduced by the use of inconsistent metrics. Looking at an example from two wetland model intercomparison studies considering several tools: Mandlazi (2017) reviewed two simulated wetlands and applied all modelling tools to the case study wetlands. However, time series analyses and FDCs were used to assess ACRU's output and paired scenarios were used to assess SPATSIM-Pitman output making it difficult to compare the wetland model's impact on streamflow across the tools. On the other hand, Maherry *et al*. (2017) reviewed five modelling tools and applied one modelling tool in one catchment. In other words, not all tools were applied to modelling a case study wetland. Thus, there was no indication of how different models could result in different predictions of streamflow volumes and regulatory roles. In terms of analysing the model output, model performance was compared to previous modelling for the same site from older studies without a wetland. Although, both studies used the paired scenario approach (i.e. comparison of flows from a scenario with and without a wetland), different metrics were used to compare the outputs from different models and in each study. This is a prime example of how different metrics makes it challenging to compare or collate the findings of the wetland models impacts on catchment streamflow from different studies.

Similar inconsistencies and challenges with metrics were reported in the analyses of global literature on wetlands streamflow regulation services completed by Kadykalo and Findlay (2016) where they reiterated the findings of Bullock and Acreman (2003): "conclusions about a wetland's flow regulation are made from different metrics from one study to the next". These studies observed that a lack of systematic characterisation and definition of wetland roles limits our ability to quantify a wetland's influence on catchment streamflow and fragments the existing knowledge. For comparing wetland models and streamflow regulation, different metrics makes it difficult to uniformly compare and identify models and their accuracy.

Considering the changing climate and model behaviour during historical events, neither of the modelling studies assessed extreme event responses. Despite the increasing frequency and severity of floods and droughts in South Africa (EMDAT-CRED, 2020), the modelled hydrological impact of wetlands during floods and droughts was not specifically assessed in these studies. Model behaviour for different climate events was not the focus of either of the studies nor was it considered for context.

### 4.1.4. Aims and objectives

Considering the absence of a multi-wetland model comparison with uniform metrics and focus on model behaviour during floods and drought, this chapter aims to identify and compare how three wetland models predict streamflow and the regulation of flows associated with an unchanneled valley-bottom wetland.

Previous studies have shown the importance of model performance based on the replicating catchment streamflow and wetland water balance variables which will considered in this study. A standardised approach for comparing simulated wetlands requires selecting the same metrics for each model's assessment. Although the output of paired scenario modelling partially depends on the land use replacing the wetland in the scenario without a wetland, the approach can be applied using any modelling tool and provides a time series of how the wetland component affects streamflow. The paired scenario approach is also open to water balance explorations. Any variable of interest (i.e. groundwater flows, interflows, AET, peak streamflow, etc.) can be assessed to identify causes for the changing streamflow predictions. In terms of providing a time series, this is beneficial for identifying temporal variability of streamflow regulation (e.g. a wetland will not always attenuate or supplement flows and the volumetric extent changes over time).

In terms of wetland model selection, three simulated wetlands from the qualitative analysis were assessed based on the modelling tools' extensive use in South Africa. The next sections will present the modelling and wetland representation for the case study wetland using ACRU's riparian zone and wetland HRUs, and the comprehensive wetland module from WRSM-Pitman. ACRU and WRSM-Pitman are locally developed models that have shown potential to conceptually represent palmiet wetlands (Glenday 2019; Rebelo, 2012, WR2012). In addition to this applicability, the models maintain a long-standing relationship with academics and practitioners. Several universities use these tools in their curriculum, and they have both been commissioned at a national level for water resource management. ACRU was instituted as the reference model for estimating the impacts of streamflow reduction activities nationally in 2008 (Jewitt *et al*., 2009) and WRSM-Pitman was commissioned as the reference tool for calculating national water resources in 2012 (Bailey and Pitman, 2016).

In doing so, the quantitative analysis addresses the final objectives for the research:

- Determining if these models differ in their predictions of the regulatory role of the Kromme wetland during extreme events: floods and droughts
- Identifying the overlaps and differences in the information gained through the qualitative assessment of wetland model structures and through the quantitative assessment of model outputs

## 4.2. Methodology

The following section describes how the wetland model comparison and streamflow regulation assessment was conducted.

### 4.2.1. Site description

#### 4.2.1.1. Locality and hydrology

The quantitative analysis focuses on modelling the catchment in which the case study wetland is located with different wetland models. The case study wetland is situated in the upper Kromme River catchment (K90A) within the Eastern Cape province of South Africa, and part of the Fish-Tsitsikamma water management area (Figure 15). The Kromme river drainage length is approximately 100 km in length, beginning in the upstream mountains and draining to the estuary at St Francis Bay which leads into the Indian Ocean.

The Kromme River catchment is a critical provider of local water resources. Listed as part of the Tsitsikamma strategic water source area, this mountainous headwater catchment receives relatively high rainfall and generates more runoff than surrounding regions (Nel *et al*., 2013). The middle and lower reaches of the catchment have two large dams: the Kromme (K90B) and iMpofu (K90D). Downstream users in Port Elizabeth depend on the Kromme catchment for 40 % of the potable water supply (Rebelo *et al*., 2015). These dams also assist with reducing flood damage on agricultural fields in downstream areas.

The upper Kromme River catchment has received significant research and development interest over the years. In addition to the construction of the previously mentioned dams, a provincial road, the R62, runs through the catchment forming an important transport route for the agricultural supply chain and tourist route connecting Cape Town and Port Elizabeth. From the early 2000s to date, efforts have been made to map and establish the local groundwater and geology dynamics (Jia, 2007; Tanner *et al*., 2019), implement erosional control structures and rehabilitate the valley-bottom wetlands (Working for Wetlands, 2005 and updates), classify the integrity and services provided by the wetlands (Ellery and Kotze, 2009, Rebelo *et al*., 2017, 2019), clear alien vegetation (Meininger and Jarmain, 2009; Hobbs, 2004; van Wilgen *et al*., 2012; McConnachie *et al*., 2012) and monitor climate variables, streamflow and boreholes supported by SAEON initiatives. See Tanner *et al*. (2019) and Cornelius *et al*. (2019) for summaries of monitoring conducted in the catchment. The case study area is a subcatchment at the head of the Kromme River that drains 49.28 $km^2$ and is monitored at the outlet of Kompanjesdrift wetland below the R62 bridge. Therefore, information and knowledge available for the site makes it a suitable location for modelling and continued research.

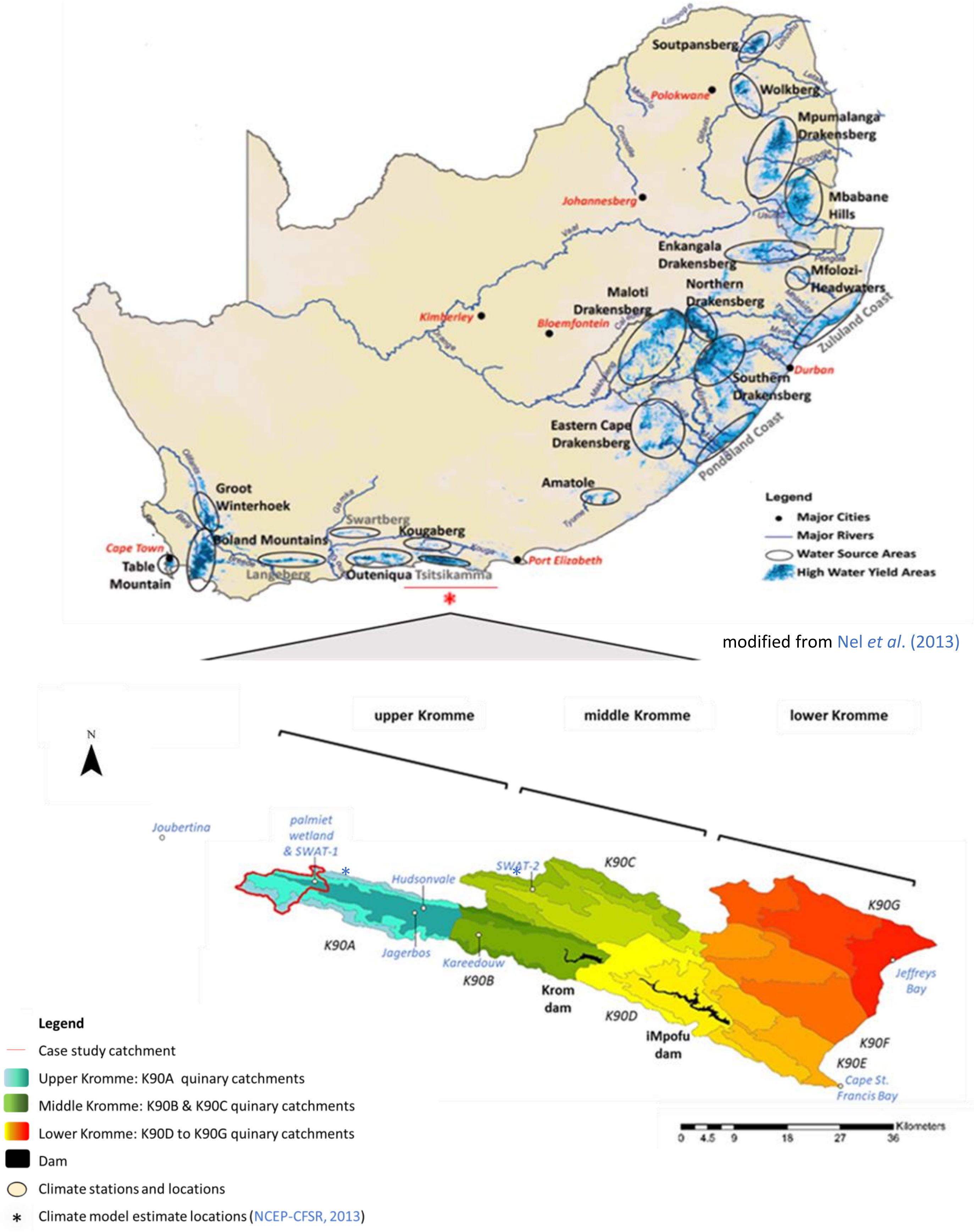



*Figure 15. Case study catchment locality*

#### 4.2.1.2. Climate

According to the Köppen-Geiger climate classification, the case study catchment is classified as a warm, temperate climate that is fully humid with warm summers (CSIR, 2015; Conradie, 2013). The aridity index classifies the site as a dry to moist subhumid area. National databases report MAP values of 716 mm p.a. for the K90A catchment (WR, 2012), 489 - 714 mm p.a. for the upper K90A catchment (Lynch, 2003) and 723 – 787 mm p.a. for the K90A quinaries (QCDB). Before spatially interpolating the climate station data and deriving a catchment average, the local weather stations and rainfall gauges (located in Joubertina, Jagerbos, Hudsonvale and Kareedouw) estimated an MAP of 620 mm p.a. from records dated October 1979 to September 2020. Bounded by the Tsitsikamma mountains to the south (peak at 1 500 m.a.s.l) and Suraanys mountain range to the north (peak at 1 050 m.a.s.l), rainfall differs on either side of the valley with higher rainfall to the south sections of the catchment. In the context of the wet east to dry west gradient, the case study catchment is located in the aseasonal rainfall zone and borders the summer rainfall zone of South Africa (Rutherford and Mucina, 2006; Mahlalela *et al*., 2020).

#### 4.2.1.3. Land cover

The Kromme catchment is within the fynbos ecoregion (Rutherford and Mucina, 2006). The majority of the case study area, upper Kromme, is in relatively pristine conditions. . Table *5* and Figure 16 presents the land cover and use properties in the case study catchment from mapping conducted by Cornelius *et al*. (2019). Agriculture is the main land use (2.25 km$^2$) while invasive wattle (1.89 km$^2$) and pine (2.09 km$^2$) trees are significantly encroaching waterways and threatening native vegetation species (fynbos and riparian woodlands of 40.18 km$^2$ and 1.21 km$^2$, respectively). Minor farm dams and direct pumping of water from the river source irrigating cultivated and pastoral lands. Three types of wetlands account for 3 % of the catchment area: palmiet peatlands in the valley, slopes and seeps.

#### 4.2.1.4. Topography and soils

The catchment elevation ranges from 1 316 m.a.s.l in the mountainous slopes to 359 m.a.s.l. at the lowest part of the wetland (Figure 17). Steep slopes bear thin soils grading into rocky media while plateaus and valleys have deeper soils. Catchment soils are mostly loamy (*QCDB*). Soil coring transects in the valley and wetland revealed that soil textures are sandy clayey and finally underlain by bedrock (Lagesse, 2017; Pulley *et al*., 2018).

*. Table 5. Upper K90A land cover and use*

| Land use | Area (km$^2$) |
|---|---|
| **Catchment (total)** | **49.20** |
| **Agriculture (total)** | **2.51** |
| **Irrigated fruit farming** | 0.53 |
| **Apple orchard (micro-spray)** | 0.24 |
| **Apple orchard (drip)** | 0.12 |
| **Stone fruit (micro-spray)** | 0.17 |
| **Irrigated pasture** | 1.65 |
| **Kikuyu/alfalfa* pasture (sprinkler)** | 1.36 |
| **Lucerne (sprinkler)** | 0.29 |
| **Minor farm dam** | 0.09 |
| **Fallow** | 0.24 |
| **Natural land cover (total)** | **41.19** |
| **Fynbos (mountain)** | 28.42 |
| **Fynbos (cliff)** | 8.70 |
| **Fynbos (lowland)** | 3.06 |
| **Riparian woodland and forest** | 1.21 |
| **Wetland** | **1.50** |
| **Palmiet** | 0.82 |
| **Slope** | 0.46 |
| **Seeps** | 0.22 |
| **Invasive plants (total)** | **3.98** |
| **Wattle (mostly riparian)** | 1.89 |
| **Pine** | 2.09 |
| **Residential (total)** | **0.02** |
| **Suburban** | 0.01 |
| **Urban** | 0.01 |

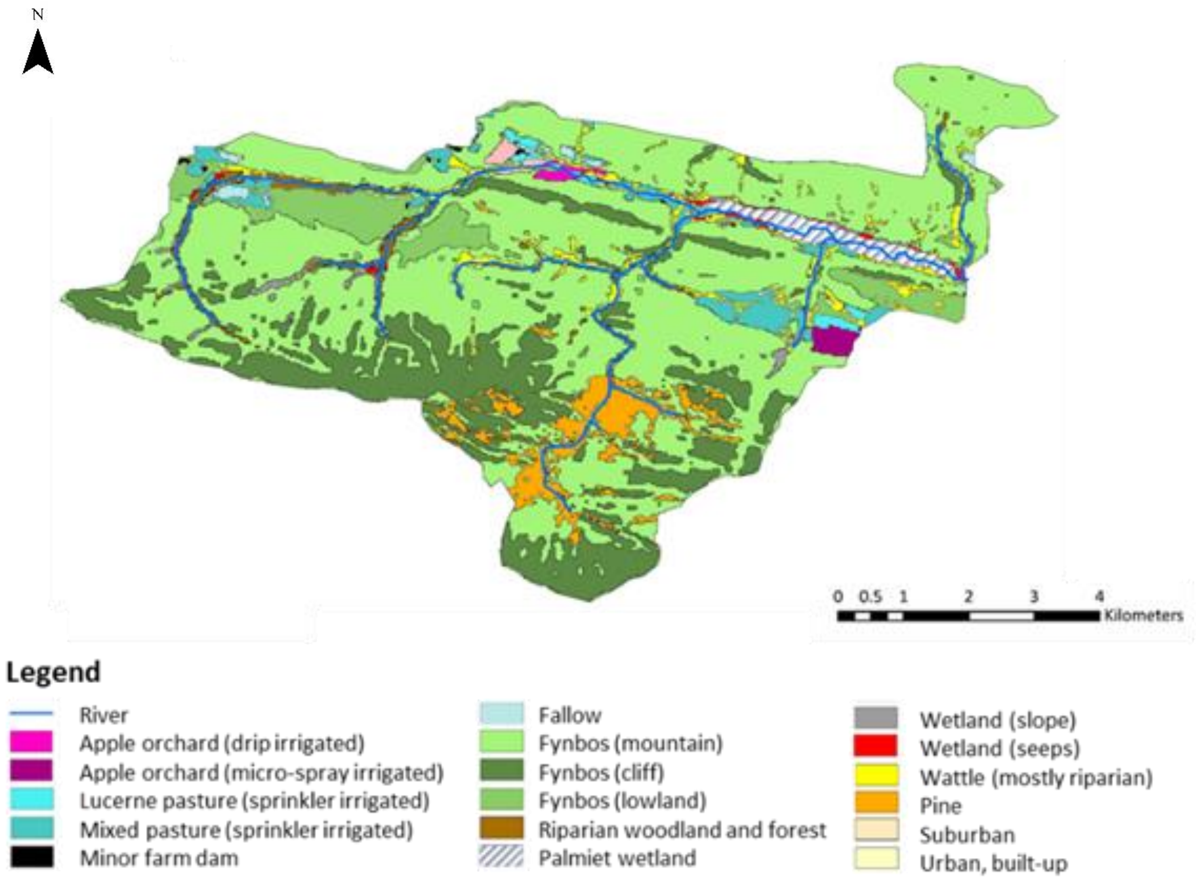



*Figure 16. Case study catchment land use and cover*

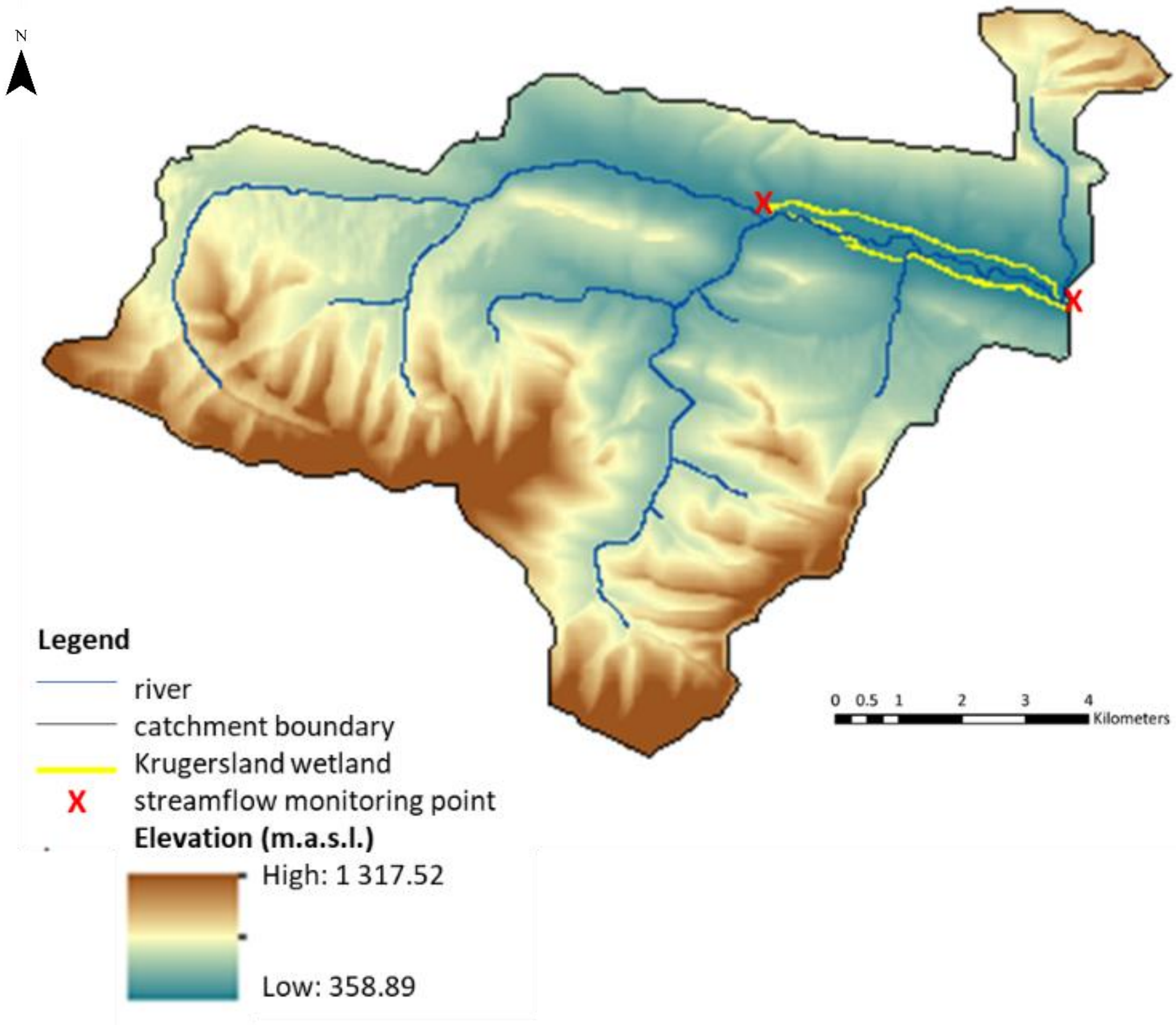



*Figure 17. Case study catchment elevation and wetland*

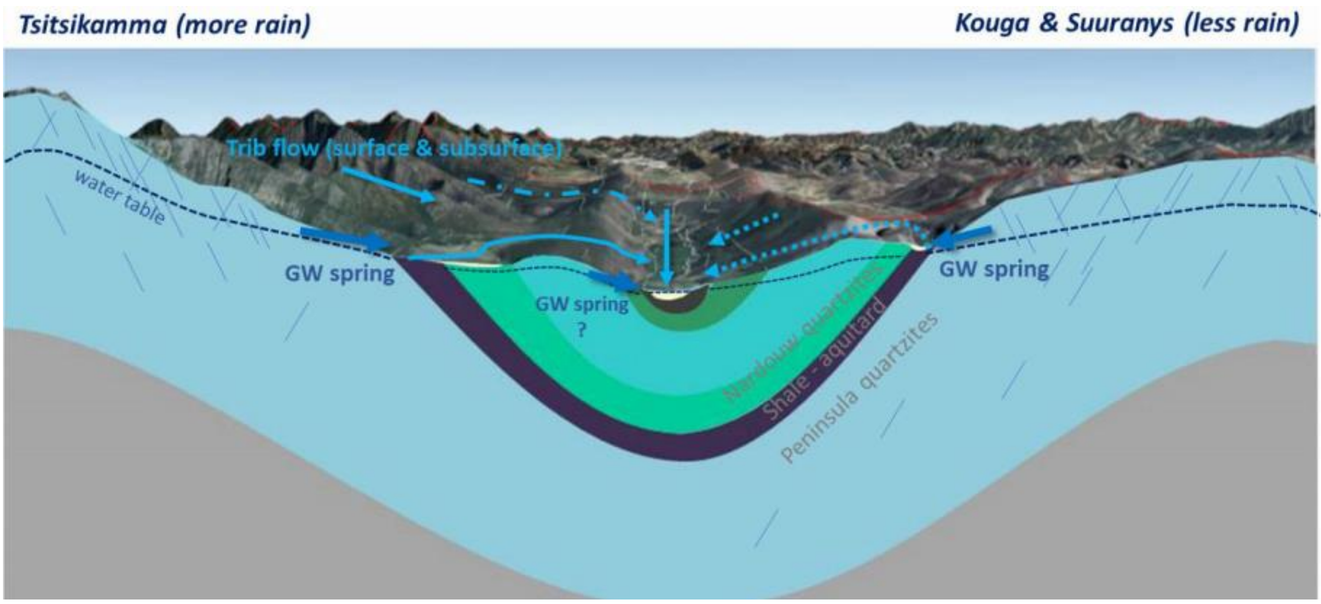



*Figure 18. Conceptual geological model in the Kromme catchment (Cornelius et al., 2019)*

#### 4.2.1.5. Geohydrology

Geological models of the of the catchment suggest that area has a large groundwater reserve which can be complicated to track or access due to complex arrangements of formations and faults. The area belongs to the regional Table Mountain Group (TMG) aquifer. The extent of fracturing and weathering influences the porosity, permeability and flow paths which determines the yield from the TMG. High yield Peninsula (mostly quartzitic sandstone with high porosities and permeability from fractures) and Nardouw (mostly quartzitic sandstone with generally lower porosities and permeability due to silt content and weathered feldspar clogging fractures) subgroups are layered between restrictive, Cederberg and Bokkeveld shale stratums to form confined aquifers, while folding and erosion have resulted in aquifer outcrops at different points in the landscape (Jia, 2007). This results in groundwater outflows from the outcrops forming tributaries and rivers over resistant materials and intermittent preferential flow pathway. The two mountain ranges bordering the main river network of the case study catchment form anticlines into the valley where there is a central syncline as illustrated in Figure 18 conceptually and from mapping in Figure 19 and Figure 20.

Regionally and within the case study quaternary catchment, the TMG geology is estimated to hold and contribute significant water volumes to runoff. According to Jia (2007), confined aquifer volumes exceed unconfined, outcrop volumes in the TMG. The total regional storage capacity of Peninsula (2.4 x $10^8$ $Mm^3$) is greater than Nardouw (7.8 x $10^7$ $Mm^3$), across full extent of these formations. Similarly, the available storage capacity of Peninsula exceeds Nardouw (2.2 x $10^6$ $Mm^3$ and 1.7 x $10^6$ $Mm^3$, respectively). Other sources quantify the groundwater contribution from the subgroup to annual runoff across the TMG region as ranging from 7.5 $Mm^3$ p.a. (Xu *et al*., 2009) to 12.2 $Mm^3$ p.a. in the national groundwater database (DWAF, 2006). With an MAR of 30.42 $Mm^3$ p.a. in K90A catchment (DWAF, 2006), the previously mentioned studies modelled groundwater contributions as 39.9 % (Xu *et al*., 2009) and 24.7 % of the catchment runoff (DWAF, 2006), respectively.

Alluvium in the floodplain and surrounding the case study wetland is considered as an additional groundwater reserve. Erosional cycles have built up alluvial deposits in toe slopes and valleys which act as superficial groundwater reserves (Pulley *et al*., 2018; Job, 2014; Grenfell *et al*., 2020; Rosewarne, 2002; Brown *et al*., 2003). Previous research in the Hex River valley, a TMG catchment in the Western Cape, supported the possibility for alluvial aquifers to contribute release large volumes of water with an estimated 5 $Mm^3$ p.a. of water seepage from alluvium to TMG subgroup aquifers (Rosewarne, 2002). In the case of the Kromme catchment, the alluvial outflows feed the palmiet wetland.

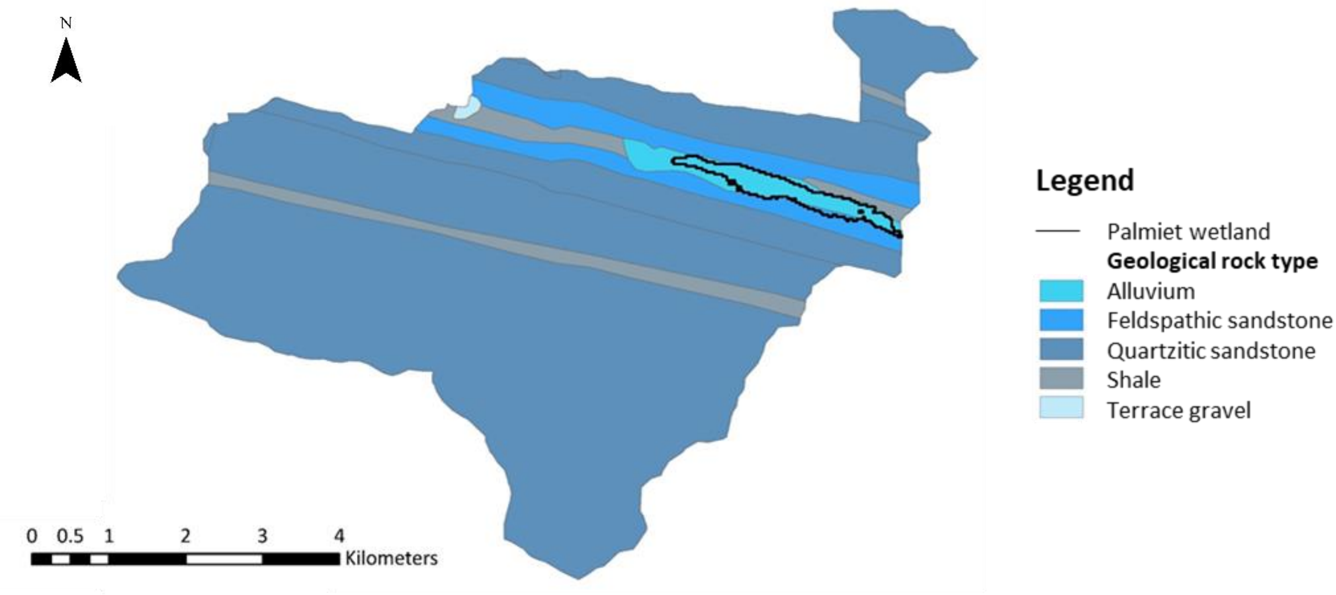



*Figure 19. Geological rock distribution in the case study catchment (modified from GCS, 2016)*

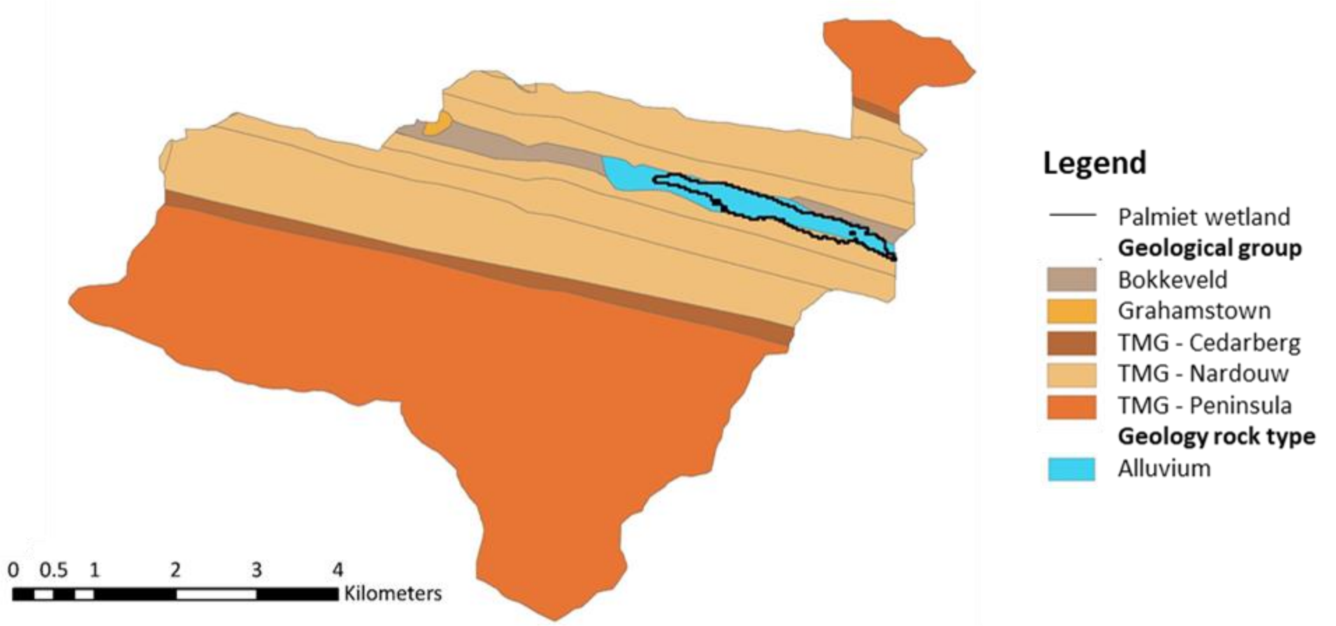



*Figure 20. Geological group distribution in the case study catchment (modified from GCS, 2016)*

### 4.2.2. Data sources and preprocessing

Table 6 presents the climate, surface and subsurface data collected and collated into input for the hydrological models. For model performance assessments, streamflow and AET data were obtained to verify the model output.

Climate data input for the modelling tools was prepared from climate stations surrounding the upper Kromme case study catchment (cf. Figure 15). The climate station in Joubertina, 20 km East of the case study site, has the most comprehensive set of variables monitored (daily rainfall, minimum and maximum air temperatures, humidity and wind speeds). Joubertina's recent records from 1997 to 2020 were combined with historical data measured within the Kromme catchment at Kareedouw (1979 to 1997).

*Table 6. Input data used to setup WRSM-Pitman and ACRU for the upper K90A case study catchment*

| Data | Description | Duration | Source |
|---|---|---|---|
| **Rainfall** | Daily | 1997 - 2020 | SAWS |
| | Daily | 1979 - 2014 | Cornelius et al., 2019 |
| **Air temperature** | Daily minimum and maximum | 1997 - 2020 | |
| | | 1979 - 2019 | SAEON |
| **Windspeed** | | 1997 - 2020 | SAWS |
| **Relative humidity** | | 1997 - 2020 | SAWS |
| **Streamflow** | Water levels at the wetland outlet | 2016 - 2019 | Glenday and Tanner, 2020 |
| **Evapotranspiration** | AET from satellite imagery | 2017 - 2020 | FruitLook, 2011 |
| | AET from scintillometer monitoring | 2019 | Tanner, 2019 |
| | Wetland water use coefficients | | Rebelo, 2012 |
| | | | Rebelo *et al.*, 2019b |
| **Land use and cover** | Distribution | 2016 | Cornelius *et al.*, 2019 |
| **Soils** | Texture, depth and porosity | - | Lagasse, 2017 |
| | | | Pulley *et al.*, 2018 |
| | | | Schulze and Horan, 2007 |
| | Vegetation rooting depth | | Richards *et al.*, 1995 |
| | | | Higgins *et al.*, 1987 |
| **Topography (DEM)** | 10 m product derived from 90 m SRTM data | - | Van Niekerk, 2016 |
| **Geology** | Distribution | - | CGS, 2016 |
| | Catchment and wetland geological model | | Tanner *et al.*, 2019 |
| | | | Jia, 2007 |

The final climate data set extended from 1979 to 2020. This covered the minimum requirement of 30 years of data for making climate-related analyses and conclusions. Windspeed and humidity data prior to 1997 when the Joubertina records begin were infilled with the day of year averages. Missing temperature data was infilled with the long-term monthly average from the Joubertina station. Solar radiation data was estimated from temperature data using Hargreaves radiation formula (*Allen et al., 1998*) and calculated in python using the pyETo package (*Richards, 2015*).

#### 4.2.2.1. Rainfall

The rainfall data received the most careful and intensive preprocessing since hydrological models are highly sensitive to rainfall inputs.

Rainfall validation (i.e. detecting missed rainfall events) and missing values were resolved based on the relationship between the Joubertina records and rainfall measured within the Kromme catchment (i.e. Kareedouw, Jagerbos and Hudsonvale). Regression analyses showed a higher linear correlation for the latter stations for records from 2017 to 2019, data prior to 2017 was based on relationships to other stations (see Cornelius et al., 2019). The regression analysis between Joubertina and Kareedouw was the same for records from 1997 to 2019 and 2017 to 2019. Missing rainfall data was infilled with the weighted average rainfall depth from Jagerbos and Hudsonvale. Potential rainfall events undetected in Joubertina were found by assuming that if it rained in Kareedouw and Jagerbos or Hudsonvale, it was likely to have rained in the case study catchment.

Spatially averaged rainfall for the upper Kromme was derived from spatial interpolation methods (i.e. inverse distance weighting, kriging and co-kriging with elevation) yielding the lowest error statistics. Long-term monthly totals for rainy and dry season months were input into the interpolation. Various literature sources indicated a bimodal rainfall distribution in the Kromme catchment (Nsor and Gambiza, 2013; Tanner *et al*., 2019). High rainfall months common in the literature were used for estimating spring/autumn rainy seasons (e.g. April and October) and low rainfall months for the dry season rainfall factors (e.g. June and July). The former, high rainfall factor was applied to all months outside of the dry season (i.e. all months excluding June and July). Kriging had the lowest error in the auto cross-validation performed in ArcGIS Pro (see **Appendix 1**). Rainfall factors from the Kriging were derived using the areal reduction factor calculation outlined in Mineo *et al*. (2018), which was the ratio of the in-situ rainfall relative to Joubertina's rainfall for the wet and dry months.

Rainfall was further interpolated into spatially representative values according to the orographic influences. Figure 22 illustrates the uneven distribution of rainfall in the Kromme catchment between the North and South sides of the valley (Glenday *et al*., 2021; Lynch, 2003). Zooming into the case study catchment in the upper Kromme, the same North-South divide of rainfall is demonstrated but it is not precisely divided by the valley (Figure 21) (Schulze, 2007; Lynch, 2003). Therefore, land uses were lumped by their spatial extent in high or low rainfall zones (i.e. North or South). The spatially averaged rainfall was then adjusted by a South and North rainfall factor. The rainfall factor was estimated as the ratio of the spatially averaged MAP from the data relative to the climate atlas MAP average for the rainfall zone (i.e. blue or yellow-brown zones in Figure 21 classified into two discrete classes in ArcGIS).The average rainfall and temperature input into the models is presented in Figure 23.

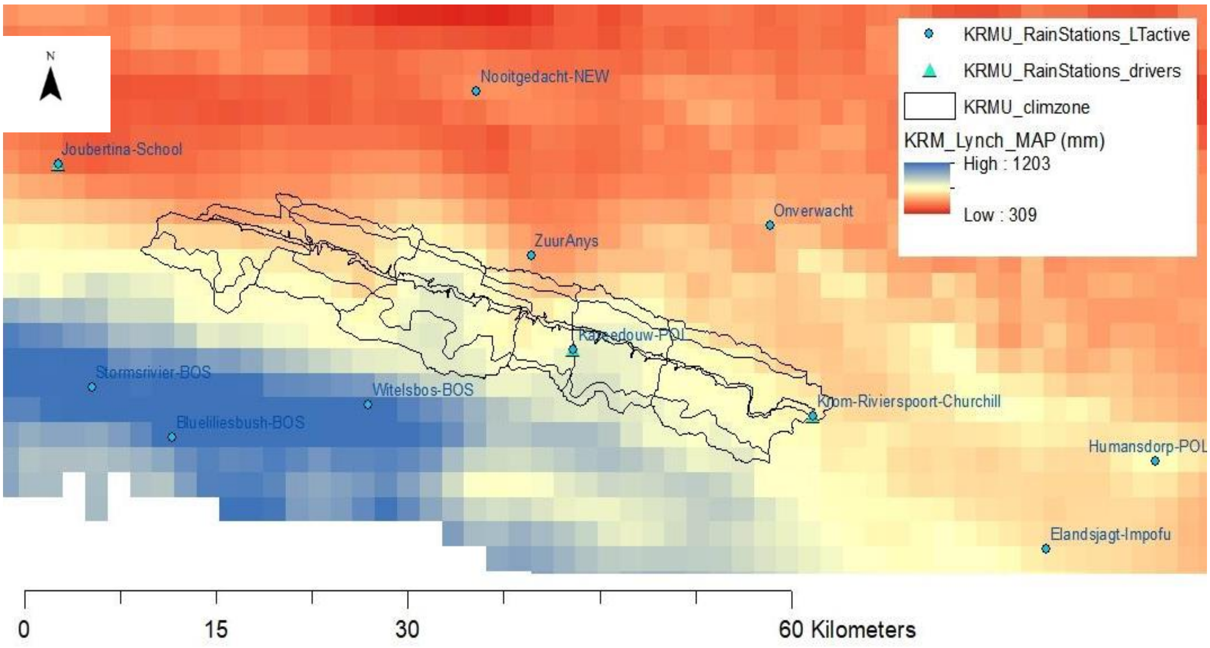



*Figure 22. Kromme catchment rainfall distribution estimated from Lynch (2003) (Glenday et al. 2021)*

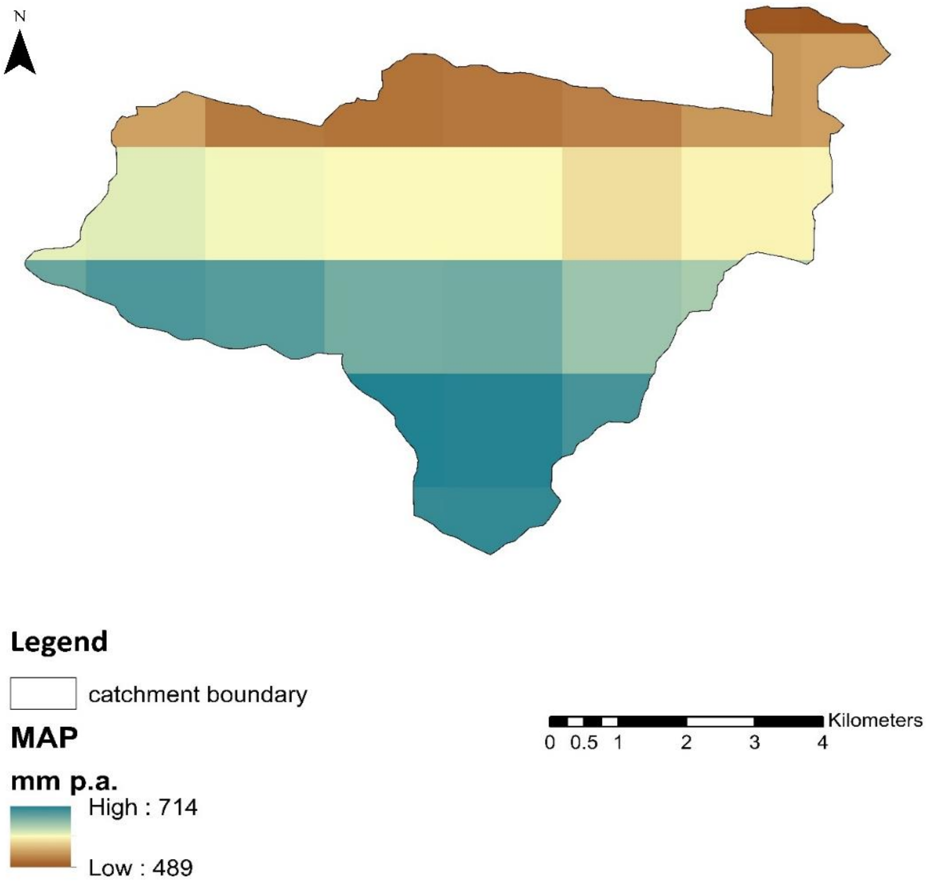



*Figure 21. Spatial distribution of rainfall in the case study catchment estimated by Lynch (2003)*

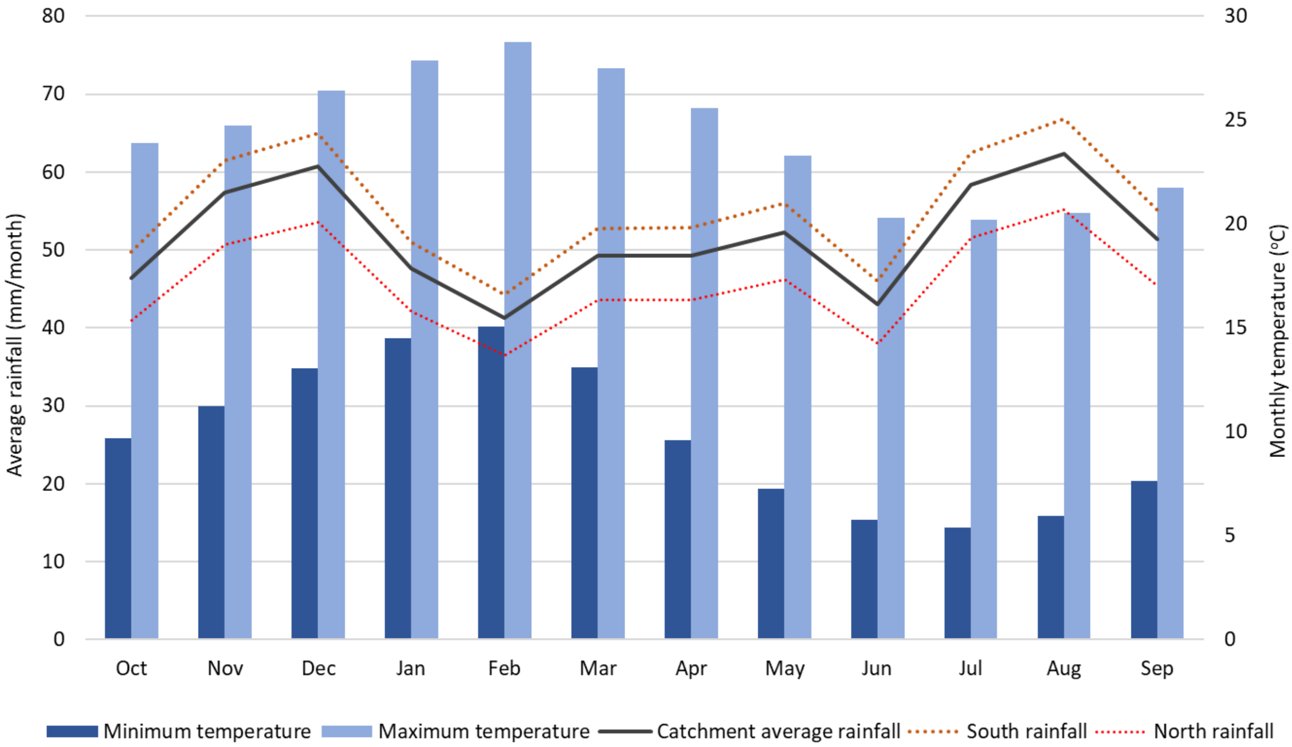



*Figure 23. Rainfall and temperature for the case study catchment during the assessment period*

#### 4.2.2.2. Evapotranspiration

Evapotranspiration datasets were derived from climate data and remotely sensed products. Reference potential evapotranspiration was calculated according to the FAO-56 method using the python package pyETo (*Richards, 2015*). AET for the case study wetland was made available courtesy of FruitLook (*FruitLook, 2011 and updates*). The AET provided is based on the SEBAL model that uses remotely sensed data, meteorological data and vegetation properties to complete the energy balance equation (Bastiaanssen *et al.*, 2005). Eight-day estimates were summed into monthly totals.

#### 4.2.2.3. Streamflow

Streamflow data was calculated from water level measurements recorded by a pressure transducer at the wetland outlet, under a bridge, positioned 0.60 m above the deepest point in the channel cross section. Rating curves to estimate streamflow from the water level data were derived based on Manning's equation and topographic survey cross sections (Glenday & Tanner 2020, per comms). Manning's roughness coefficient and slope values, and their relationship to water depth, were adjusted improve the fit of calculated streamflow to manual flow measurements. Manual flow measurements could only be taken in low to medium flow conditions. As such, the timeseries of flow in to the Krom Dam (DWS) was used as an additional reference, particularly for high flows, rescaling it by the ratio of the case study catchment area to the dam's catchment area. The resulting rating curve was used to generate a streamflow time series for the roughly three years of pressure transducer data, from June 2016 to October 2019. Flows when the water level dropped below the pressure transducer were assumed as 0.01 $m^3 s^{-1}$. Extremely low flows down to stagnant pools and dry channel conditions were observed at these levels during site visits, but it is likely that the river was not always completely stagnant or dry during these periods. Despite these limitations, the data record correlates well with water levels measured at the wetland section immediately downstream of the case study wetland. Flow rates were converted to depths using the upstream catchment area.

### 4.2.3. Tool configuration

In this section, schematic diagrams for the modelling software's catchment assumed that surface water represents fast runoff and surface runoff, groundwater represents baseflow, groundwater outflows; and interflow represents slow or delayed runoff components from the soil.

#### 4.2.3.1. ACRU model description

The ACRU modelling tool is a physical-conceptual, multi-layered soil water budgeting, daily time step model (Schulze, 1995 and updates). ACRU's catchment is distributed into a set of land units, each of which represent areas that respond similarly to rainfall inputs with similar runoff generating mechanisms. These units are called hydrological response units (HRUs). Each HRU has an explicit surface area, land cover, and subsurface storages. A conceptual diagram of the model algorithms for HRU processes and storages is presented in Figure 24. If there are any impervious areas specified within a subcatchment, the first portion of runoff is generated from these areas. Thereafter, water is directed to the soil profile of HRUs for soil water budgeting.

Similar processes and storages occur within and from each HRU. As a semi-distributed model, ACRU differentiates the catchment into subcatchments with HRUs. Daily rainfall inputs are assigned to the HRUs. First, rainfall is intercepted by the vegetation. A fraction of the rainfall depth is allocated to interception at a vegetation-specific rate. Stored interception water is the first source for daily ET demands before the soil moisture in the root zone storage. Rainfall in excess of interception thresholds infiltrates the soil based on the soil texture and antecedent wetness (within the topsoil or user-specified runoff generating depth of the soil). For the throughfall (i.e. water not intercepted by the vegetation) that does not infiltrate, user-defined parameters determine the proportion that becomes runoff on the same day (termed as quickflow in the tool) and runoff leaving the HRU in days after the rainfall event (referred to as delayed stormflow). In the conceptual diagram, this constitutes the surface water runoff from the HRU and is referred to as fast and slow runoff, respectively. In this study, fast runoff was assumed to equate to surface runoff and slow runoff represented interflow.

The root zone storage for a HRU is represented as a soil profile with a topsoil and subsoil. Soil moisture storage in excess of the field capacity drains from the upper soil profile (i.e. topsoil or A-horizon) to the lower soil profile (i.e. B-horizon or subsoil). Soil water draining from the B-horizon is routed to groundwater storage. The second portion of runoff, groundwater outflow, is generated from the groundwater storage unit which is below the root zone storage. The capacity of the groundwater storage is limitless. Groundwater outflows are calculated as a percentage of the current volume. In terms of evapotranspiration, parameters describing the static vegetation properties above and below the ground determine the vegetation water use from the soil profile which is temporally variable (viz. ET responsive to the daily soil moisture and climate conditions).

In terms of wetland models in ACRU, specialised HRUs with different routing allowances are used. General HRUs route outflows in parallel to the main river channel (i.e. water moves from the HRU to the river). However, the wetland and riparian zone HRU representing wetlands in ACRU allow for alternative flow routing options. These HRUs simulate overbank spilling using a river threshold capacity. Through this overflow, the wetland and riparian zone HRUs receive inflow from the upstream subcatchment or HRUs. This channel overflow input includes surface runoff and groundwater from the upslope HRUs. The inflowing water is added onto the surface of the wetland or riparian zone HRU where it may infiltrate or become

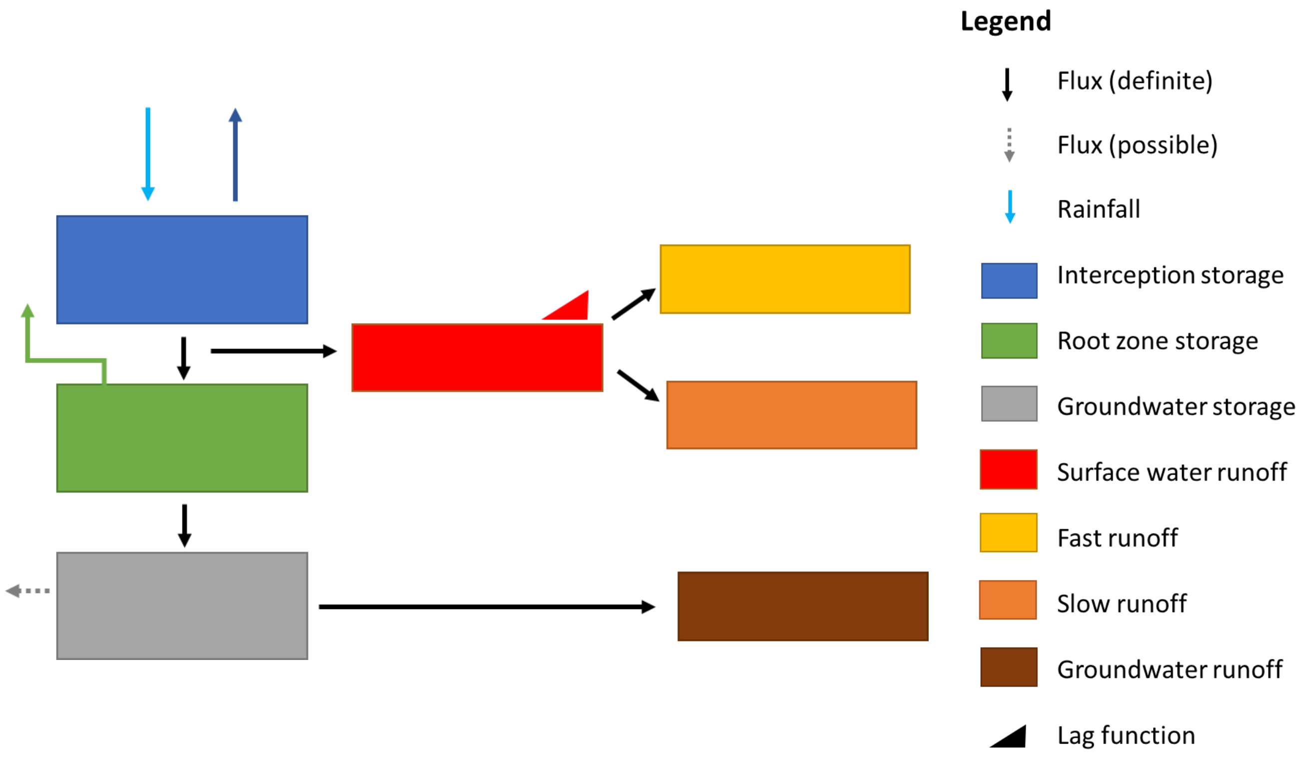



*Figure 24. Concept for basic runoff generation unit in ACRU*

surface runoff. In the riparian zone wetland HRU, a fraction of the upstream groundwater from upslope HRUs in the same subcatchment can be routed into its subsoil (i.e. subsoil of the root zone storage). Subsoil inflows to the riparian zone HRU were implemented to represent riparian zone vegetation's access to groundwater. A parameter specifies the proportion of upstream groundwater routed to the subsoil of the riparian zone HRU. Further details describing the ACRU wetland models are presented in Section 3.3.1.1 and 3.3.2.

#### 4.2.3.2. WRSM-Pitman model description

WRSM-Pitman is a conceptual, modular, monthly time step modelling tool that simulates the movement of water through interlinked system of catchments or subcatchments, river reaches, reservoirs, irrigation fields and mines (Bailey and Pitman, 2016). The catchment, or each subcatchment, is represented with a "runoff module" which receives rainfall (Bailey and Pitman, 2015; Figure 25). The first portion of runoff is generated from impervious areas, as immediate runoff, if an impervious area has been specified within the catchment area. Rainfall is then intercepted by the vegetation. The second portion of runoff (represented as surface water in the conceptual diagram) is generated based on infiltration rates under dry and wet conditions, and the soil moisture in the root zone storage (referred to as the soil zone in the tool). Surface runoff is generated when the soil is saturated, and when rainfall exceeds the infiltration capacity for the month. The final volumes of runoff are generated from the unsaturated and saturated zones of the soil, or subsurface profile, as interflow runoff and groundwater outflows.

Using the Sami groundwater routine, water can percolate from the root zone storage into an intermediate, unsaturated zone preceding the groundwater storage. Interflow is generated when the soil moisture in the root zone storage is above a minimum soil water retention value and below the maximum storage. When the intermediate layer is saturated, the excess water moves into the underlying groundwater aquifer. Interflow is also generated when the intermediate, unsaturated zone and groundwater storage are saturated. These two methods of interflow runoff generation are referred to as slow runoff in the conceptual diagram.

The saturated zone generates groundwater discharge depending on the groundwater volume above a user-specified threshold which initiates outflows and the aquifer transmissivity. Groundwater recharge is lagged depending on the storage in intermediate, unsaturated storage zone to represent the slow drainage of water percolating from the soil to the aquifer. Water supporting ET can be sourced from the root zone and from the groundwater storage. Groundwater availability for AET is specified as a percentage of the runoff module area and related to the area of the catchment in low lying areas with access to groundwater.

Regarding the wetland model in WRSM-Pitman, wetlands are conceptualised in channel modules. The comprehensive wetland model is conceptualised as a reservoir that evaporates at a fixed rate representing wetland vegetation using pan coefficients. Similar to open water bodies, evaporation from the wetland model is assumed to occur consistently at a maximum rate until water in the wetland storage is depleted. Therefore, there is no reduction or variability in wetland evaporation as the wetland unit's water storage drops that would account for declining soil moisture, the vegetation's wilting point and interannual climate variability. In terms of streamflow regulation, the wetland module has a set of thresholds based on the carrying capacity of the upstream river and proportioning upstream flows into wetland inflows and water flowing through the wetland. Wetland outflows to the downstream river depends on the user-defined wetland storage being exceeded. Return flows to the river occur at a user-defined rate which specifies the proportion of excess storage released every month. Section 3.3.1.1 and 3.3.2 describes the comprehensive wetland model in more detail.

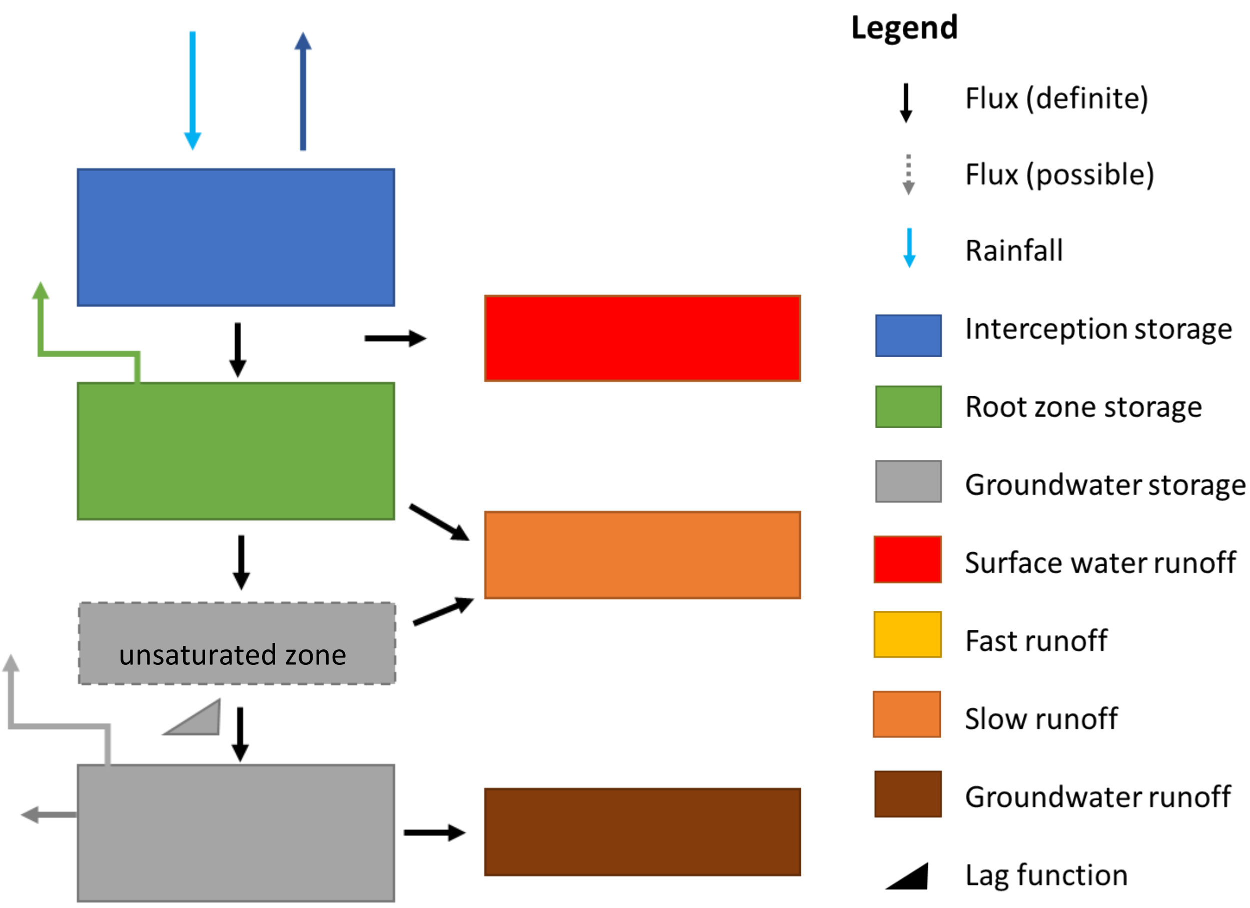



*Figure 25. Concept for basic runoff generation unit in WRSM-Pitman*

### 4.2.4. Case study configuration

Three models with the same climate input data and areas, but with different wetland routines, were setup for the upper Kromme catchment (Table 7). Areas of different land uses were categorised into the North or South section of the valley and received the associated rainfall. For all models in the scenarios without a wetland, the wetland area was allocated to North fynbos runoff module of WRSM-Pitman and North fynbos HRU of ACRU models.

In total, three different wetland routines were configured for the palmiet, unchannelled valley-bottom wetland: the first used the comprehensive wetland from WRSM-Pitman (hereafter referred to as the CW model); the second used the riparian zone HRU in ACRU4 (RZ model); and the third used the wetland HRU in ACRU4 (WL model).

*Table 7. HRUs and subcatchments areas (km$^2$) in the WRSM-Pitman and ACRU model*

| ***Land use/cover*** | ***Catchment area receiving N rainfall*** | ***Catchment area receiving S rainfall*** |
|---|---|---|
| **Fynbos** | 20.45 | 19.73 |
| **Pine** | | 2.01 |
| **Wattle** | 1.89 | |
| **Woodland** | 1.21 | |
| **Wetland** | 1.5 | |
| **Dam** | 0.0754 | |
| **Orchard** | 0.53 | |
| **Pasture** | 1.89 | |
| **Subtotal** | **27.54** | **21.74** |
| **Total** | **49.28** | |

Figure 26 presents the conceptual diagram for the catchment of the Kromme wetland case study in the WRSM-Pitman model. Figure 26, Figure 27, and Figure 28 illustrates the model specific conceptual diagram in ACRU models. Table 8 and Table 9 presents the wetland parameterisation in the WRSM-Pitman model and ACRU models. The simulated wetland concepts are outlined in detail in Chapter 3.

The CW wetland was configured as a riparian wetland with all upstream river flows entering the wetland because the Kromme case study wetland intercepts all upstream channels (Figure 26; Table 8). Perennial, subsurface preferential flow pathways in the wetland and side-cutting channels active during wet seasons were incorporated as a proportion of incoming river flows entering the wetland flowing directly through the wetland, bypassing the wetland's storage.

This was considered reasonable given the model's monthly time step. In the model, 40 % of river inflows entered the wetland storage while 60 % flowed through the wetland to the downstream river within the same month. The adjustment from having all inflows enter the wetland storage and be subject to attenuation and ET, to instead having a fraction of the inflows bypassing this, was introduced during the calibration period to account for a steep rising limb in the observed streamflow hydrograph. The wetland storage was estimated as the volume of water potentially retained within the soil profile depth accounting for the porosity of the soil. This volume was used as the nominal wetland volume in the CW wetland. The nominal wetland storge initiated the release of outflows to the downstream river when the wetland storage exceeded the nominal wetland storage. If the wetland storage was less than the nominal wetland storage, no return flows to the downstream river occurred. In addition to this storage regulation, no proportional lagging of wetland storage above the nominal storage was applied (i.e. all wetland storage above the nominal wetland storage was assumed to flow out of the wetland in the next month) (Table 8). This was assumed to be hydrologically reasonable since the physical wetland does not have ponded water on its surface. A schematic diagram of the wetland water balance and streamflow regulation was presented in Chapter 3, Figure 12.

*Table 8. WRSM-Pitman parameters used in the comprehensive wetland setup*

| Parameter | Range | | Calibrated value | Units | Description |
|---|---|---|---|---|---|
| | Min | Max | | | |
| **Area** | - | - | 1.5 | $km^2$ | Area of the wetland at bankfull level of channel |
| **Snom** | - | - | 2.43 | $Mm^3$ | Nominal wetland volume |
| **beta** | 0.2 | 0.8 | 0.4 | Fraction | Power of the area-volume relationship (calibrating scale based on shape) |
| **Qbf** | - | - | 0 | $Mm^3$/month | Bankfull capacity of the river channel |
| **Kin** | 0 | 1 | 0.4 | Fraction | Proportion of channel flow in excess of Qbf into wetland storage |
| **Kout** | 0 | 1 | 1 | Fraction | Proportion of wetland volume over Snom into river |
| **Bedloss** | - | - | 0.02 | $Mm^3$/month | Bed losses from the catchment river upstream of the wetland |

The ACRU model with the RZ wetland was setup with one subcatchment (Figure 27). Upstream surface water runoff was an inflow to the wetland's surface via the river overbank spilling. Upstream groundwater outflows were routed to the B-horizon soil of the RZ wetland. The real wetland is completely within the riparian zone. Based on this spatial extent and perennial subsurface inflows noted in the observations from Tanner *et al*. (2019), the RZ wetland was parameterised to receive all upstream baseflows in the subsoil. In this case, the subsurface water in the main river which was noted in the same study by Tanner *et al*. (2019) was not considered in the RZ wetland setup. This exclusion is partially justifiable since the distribution of runoff into subsurface and surface components in channel flow from the main river and side tributaries, and in the subsurface flow pathways are yet to be quantified along the case study wetland (observations were concentrated in the Kompanjiesdrift wetland immediately after the case study wetland). What is certain from the study is that there is TMG aquifer water contributing to the wetland inflows via the river and as subsurface contributions using electrical resistivity tomography surveys, chemical isotope tracing and mixing cell modelling (Tanner *et al*., 2019; Smith, 2019). Without an exact indication of the surface-groundwater proportions, as a start to building models for the Krom wetland, the extreme flow pathways were explored: firstly, with all upstream groundwater water entering the wetland as subsurface flow in the RZ wetland; and secondly, with all upstream groundwater entering the wetlands as part of the main river inflows in the WL wetland.

The model with the WL wetland was setup with two subcatchments (Figure 28). The first subcatchment included all upstream (i.e. non-wetland) land uses and all the surface and groundwater outflows from this area became inflows onto the surface of the WL wetland, in the downstream subcatchment, via overbank spilling. Over bank spilling was processed by setting the channel capacity to 0m$^3$/day in both wetlands (Table 9). This allowed all channel flows to spillback onto ACRUs wetlands. Infiltration was calculated based on the soil profile's soil moisture. In-field measurements of the wetland soil porosity were used to parameterise both wetland HRUs in ACRU.

*Table 9. ACRU wetland parameters used for the riparian zone and wetland HRU*

| Parameter | Range | | Calibrated value | Units | Description |
|---|---|---|---|---|---|
| | Min | Max | | | |
| **QFRESP** | 0 | 1 | 0.10 | Fraction | Proportion of stormflow leaving the HRU on the same day it is generated, or of rainfall event |
| **SMDDEP** | 0 | soil depth | 3.45 | m | Critical soil depth in which the soil moisture influences stormflow generation |
| **ARESP** | 0.1 | 0.8 | 0.40 | Fraction | Soil water drainage rate when A horizon is above field capacity (A to B horizon) |
| **BRESP** | 0.1 | 0.8 | 0.70 | Fraction | Soil water drainage rate when B horizon is above field capacity (B horizon to groundwater storage) |
| **COFRU** | 0.001 | 1 | 0.05 | % | Coefficient of baseflow response (fraction of storage that flows out per day) |
| **Channel capacity** | - | - | 0 | m$^3$/day | Flow rate above which channel flows spill onto the wetland |
| **PCRIPINFEST**[1] | 0 | 100 | 100 | % | Riparian area infestation – implemented as the percent of baseflow output from upland HRUs that is routed to the soil B horizon of the riparian zone (remainder routed to streamflow) |

[1] option in the riparian zone wetland setup only

Outflows from the ACRU wetlands were parameterised identically (Table 9). With the QFRESP set to 0.1, 90 % of rainfall after canopy interception and the soil moisture deficit in the profile were accounted for, became stormflow on the following day which was assumed to represent interflow. Water drainage rates decreased down the soil profile. Groundwater outflows were released at a rate of 5 % of the current storage volume. ACRU's output was upscaled to monthly flows for comparisons with WRSM-Pitman.

A schematic diagram of the wetland water balance and streamflow regulation was presented in Chapter 3, Figure 8 for the RZ wetland and Figure 9 for the WL wetland.

For all three models, the wetland water balances are presented in section 4.2.5.2

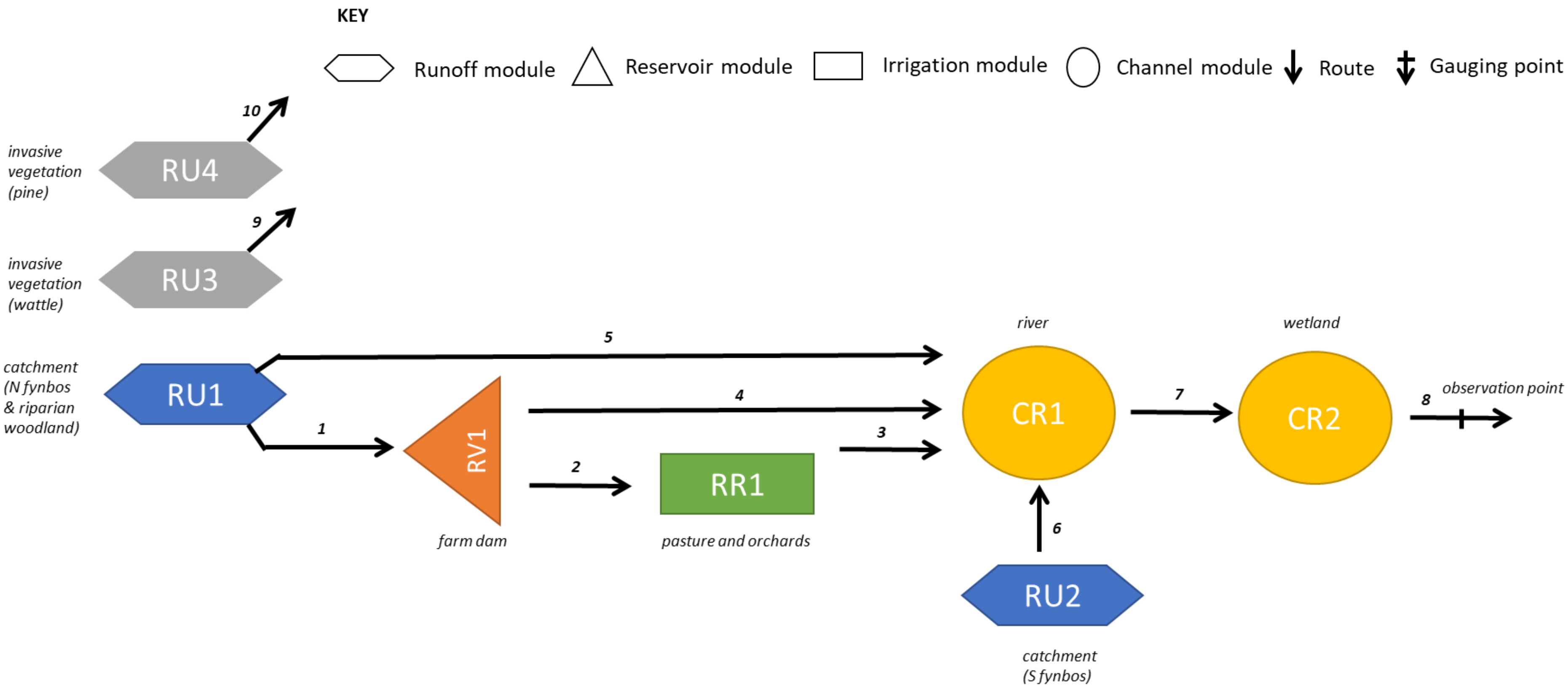



*Figure 26. Conceptual catchment model (network diagram) setup in WRSM-Pitman with the comprehensive wetland*

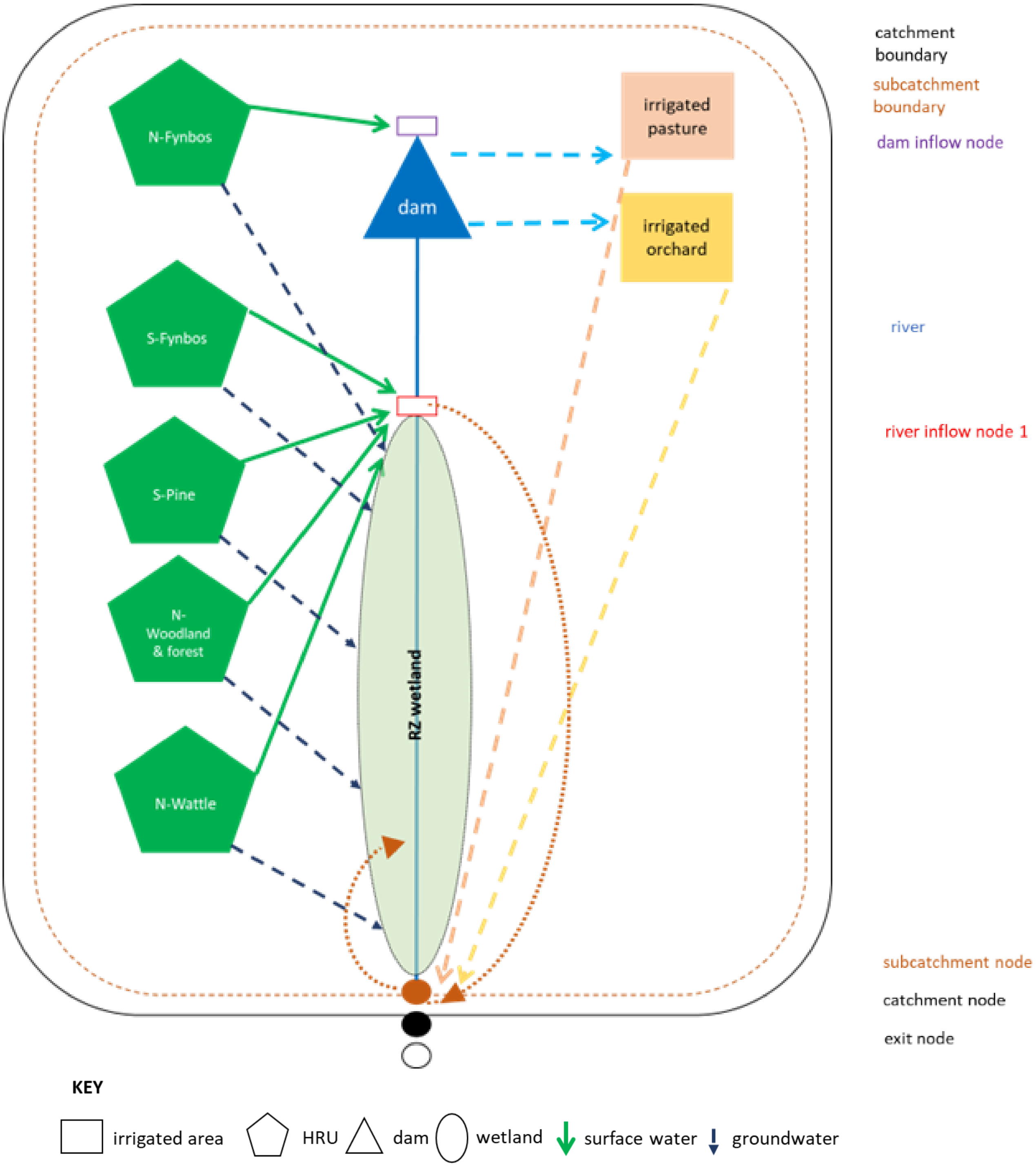



*Figure 27. Conceptual catchment model setup in ACRU with the riparian zone HRU*

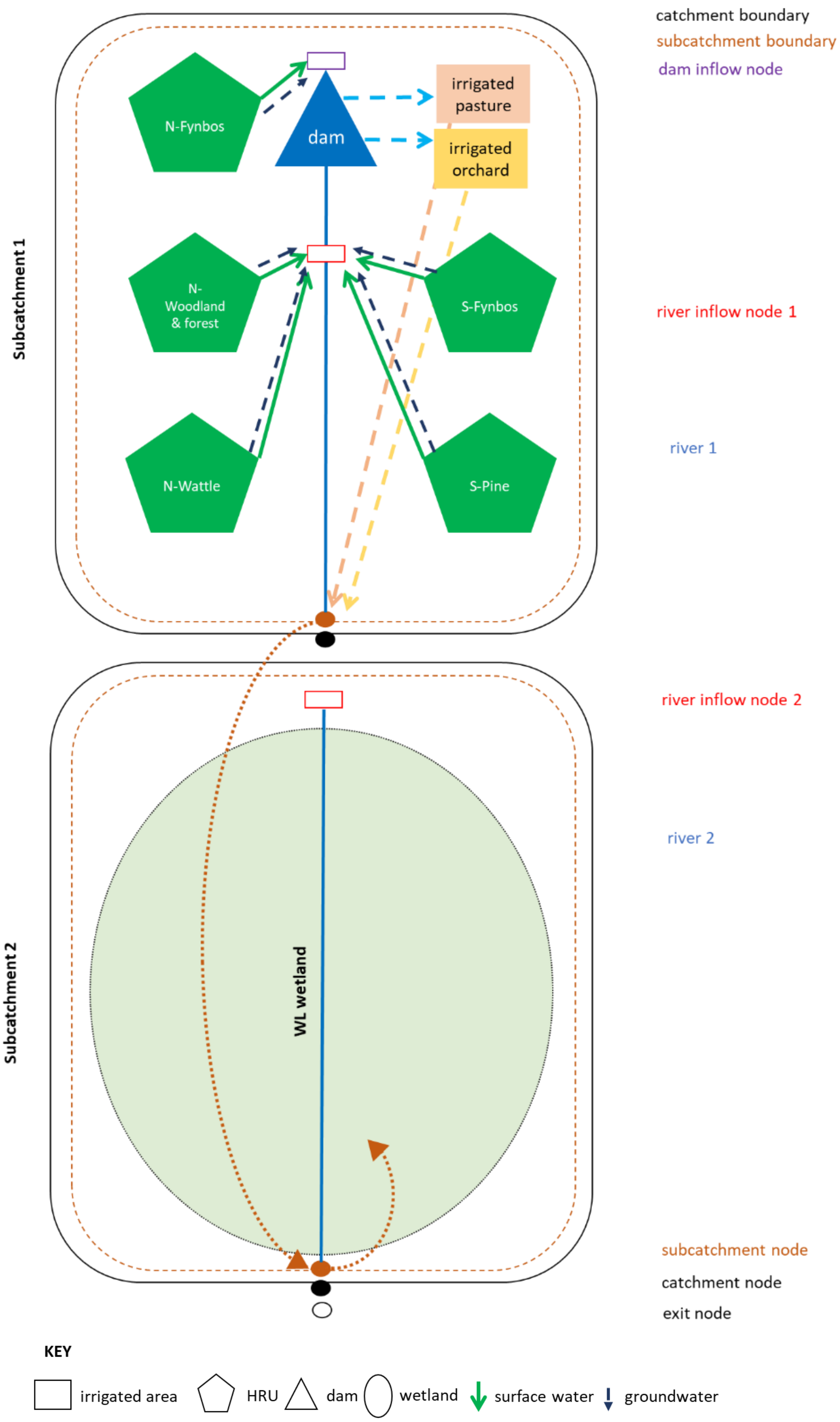



*Figure 28. Conceptual catchment model setup in ACRU with the wetland HRU*

## 4.2.5. Calibration and analyses

### 4.2.5.1. Model performance

The hydrological models were calibrated according to the streamflow records. Thereafter, calibrated models were further validated for model performance based on the AET records. Catchment parameters were the initial focus of the WRSM-Pitman model. Final parameter adjustments focused on refining the wetland concept and parameters. For the ACRU models, the RZ model was first setup and calibrated. The same parameters were then applied to the WL model setup.

Streamflow and AET were statistically assessed using a range of metrics covering all flow periods. The performance of simulated results compared to observed values were analysed using the Nash-Sutcliffe Efficiency (NSE, Nash and Sutcliffe, 1970) for high flows, Kling-Gupta Efficiency (KGE, Gupta *et al*., 2009) for average flows and log-transformed values as input into the NSE equation for low volumes. The optimal value for NSE is 1. The KGE was used as an additional goodness of fit metric (Knoben *et al*., 2019). Suboptimal performance is indicated in KGE values less than -0.41. The KGE was developed to address the shortcomings and bias of the NSE to high flows. Log-transformations of a variable in the NSE equation increases the weighting given to low values.

Equations 1 and 2 describe the calculation of the performance indices:

$$\text{NSE} = 1\text{-}\left[\frac{\sum_{I=1}^{n}(\,Yi^{obs} - Yi^{sim})^2}{\sum_{I=1}^{n}(\,Yi^{obs} - Yi^{mean})^2}\right] \tag{1}$$

$$\text{KGE} = \sqrt{(r-1)^2 + \left(\frac{\sigma_{sim}}{\sigma_{sim}} - 1\right)^2 + \left(\frac{\mu_{sim}}{\mu_{sim}} - 1\right)^2} \tag{2}$$

where $Yi^{obs}$ is the ith observation for the evaluated model, $Yi^{sim}$ is the ith simulation for the evaluated model, $Yi^{mean}$ is the mean of the observed data and n is the total number of observations.

r is the first objective in KGE representing the linear correlation between the observations and simulations; alpha is the second objective of the KGE where $\sigma_{sim}$ is the standard deviation in the observations, $\sigma_{obs}$ is the standard deviation in the simulations; beta is the third objective in the KGE metric where $\mu_{sim}$ is the simulation mean, and $\mu_{obs}$ is the observation mean.

The NSE and KGE metrics were interpreted according to the standards in Table 6 after Bennet *et al.* (2013) and Knoben *et al.* (2019). The coefficient of determination ($R^2$), variation (CV) and standard deviation (SD) were added as additional measures describing the model performance. All statistics were computed using the hydroeval 0.1.0 package (Hallouin, 2018 and updates).

*Table 10. Statistical indicators and categories of model performance*

| **Model performance** | **NSE** | **KGE** | **NSE (log Q)** | $R^2$ |
|---|---|---|---|---|
| **Range** | - ∞ to 1 | - ∞ to 1 | - ∞ to 1 | -1 to 1 |
| **Excellent** | 0.7 – 1.0 | 0.41 – 1.0 | 0.7 – 1.0 | 0.5 – 1.0 |
| **Acceptable** | 0.5 – 0.7 | > - 0.41 | 0.5 – 0.7 | > 0.5 |
| **Poor** | < 0.5 | < -0.41 | < 0.5 | < 0.5 |

AET model performance was assessed using the full data set and a percentiles dataset. The same model performance statistics used for streamflow were computed for both the timeseries data set and for the percentile distribution of the AET depths (**Appendix 2**). The percentiles datasets consisted of the $10^{th}$, $20^{th}$, $30^{th}$, $40^{th}$, $50^{th}$ (median), $60^{th}$, $70^{th}$, $80^{th}$ and $90^{th}$ percentile ET values and the minimum and maximum ET in the observation data and model output. To reduce the restrictions on model performance in accordance with the variable time scales of the model output (e.g. daily and monthly) and remotely-sensed observed data records (i.e. 8-day depths upscaled to monthly values), statistics were supported with the total, average and median flows from the full simulated data set of AET. The CW wetland AET was calculated as the product of the monthly Kc and PET, because the model never predicted the wetland water storage to drop below the ET demand. AET from ACRUs wetlands were calculated as the sum of modelled soil water evaporation and transpiration.

#### 4.2.5.2. Water balances

With ΔS representing the change in storage, the distribution of water in the catchment and wetland water balances were estimated for each model according to equations 3 – 7.

Groundwater, soil moisture and wetland storage fluxes can be increasing or decreasing at different times in the simulation period. In the following equations, all inflows are presented in regular, dark font and outflows are listed as italicised and grey font on the right-hand side of the equation:

- **The CW model catchment**

$$\Delta\ \mathbf{S\ (soil\ moisture + aquifer)} = \text{rainfall} - \textit{surface runoff - interflow - groundwater outflows - AET} \tag{3}$$

- **The ACRU catchment for both the RZ and WL models summarised from all HRUs**

$$\Delta\ \mathbf{S\ (soil\ moisture + groundwater)} = \text{rainfall} - \textit{surface runoff ("quickflow") - interflow ("delayed stormflow") - groundwater outflows ("quickflow") - AET} \tag{4}$$

- **The CW wetland in WRSM-Pitman**

$$\Delta \text{ S (wetland)} = \text{rainfall} + 0.4 * \text{river inflows} - \textit{AET - wetland return flows} \quad (5)$$

- **The RZ wetland in ACRU**

$$\begin{gathered}\Delta \text{ S (wetland)} = \text{rainfall} + \text{upstream HRU groundwater outflows routed to wetland subsoil} \\ + \text{upstream surface runoff -} \\ \textit{AET - interception - surface runoff - interflow - groundwater outflows}\end{gathered} \quad (6)$$

- **The WL wetland in ACRU**

$$\begin{gathered}\Delta \text{ S (wetland)} = \text{rainfall} + \text{upstream surface runoff} + \text{upstream groundwater outflows -} \\ \textit{AET - interception - surface runoff - interflow - groundwater outflows}\end{gathered} \quad (7)$$

All flows in each water balance were calculated as volumes ($Mm^3$). The annual average was computed for hydrological years of the calibration period (October 2016 – September 2019) and assessment period (October 1989 – September 2020). A warm-up period of three and ten years was used in each period, respectively. Since both ACRU and WRSM-Pitman equilibrate quickly, it was assumed that after the warm up period the models were assumed to have reasonable starting values. Inflows and outflows presented in the wetland water balances (Equations 5 – 7) were used in the event analyses.

### 4.2.6. Wetland impact

#### 4.2.6.1. Impact indicator

Simulated streamflow from scenarios with and without a wetland were set as the baseline indicator for the wetland components impact on streamflow. The absolute hydrological impact is the difference between the two outputs (i.e. the output variable from the model runs with and without a wetland) (Equation 8). Relative impact was expressed as a factor, calculated as the absolute impact divided by the output from the simulation without a wetland (Equation 9). A positive number indicates supplementation, while a negative number specifies that the wetland is reducing the flow in the model (i.e. attenuation). Wu *et al*. (2019) and Fossey and Rousseau (2016) describe the metric in more detail.

$$\text{Absolute wetland impact factor} = \text{Q with a wetland - Q without a wetland} \quad (8)$$

$$\text{Relative wetland impact factor (\%)} = \frac{\text{absolute wetland impact factor}}{\text{Q without a wetland}} x\ 100 \qquad (9)$$

The wetland impact factors were applied to two hydrological signatures: flow duration curves and time series. Hydrological signatures characterise the response of specific catchment processes or behaviours (Sawicz *et al*., 2011). Using hydrological signatures as a performance metric gives insights into how adequately processes are represented within a modelling tool (Pool *et al*., 2017). The selected signatures were selected to give insights into the hydrological response for each model to individual floods and extreme flows.

#### 4.2.6.2. Event detection

Extreme hydrological events were detected using the Standardized Precipitation Evapotranspiration Index (SPEI) supplemented with historical records and literature. The SPEI uses precipitation and PET as estimates of water supply and demand to delineate phases of dry and wet conditions (Vicente-Serrano *et al*., 2010). The index is divided into four categories. Positive SPEI values indicate wet conditions according to these categories: extreme (SPEI > 2), severe (1.99 > SPEI > 1.5), moderate (1.49 > SPEI > 1.0) and mild (1.0 > SPEI > 0). Negative values indicate dry conditions according to the following categories: extreme (SPEI < -2), severe (-1.99 < SPEI < -1.5), moderate (-1.49 < SPEI < -1.0) and mild (-1.0 < SPEI < 0). Floods usually occur at fine time scales of days, while droughts occur over longer periods. Because the WRSM model runs at a monthly time step, this was the smallest time step used for event selection. The 1-month SPEI (hereafter, SPEI-1) was used to detect wet conditions potentially associated with floods. The 12-month SPEI (hereafter, referred to as the SPEI-12) was used to detect hydrological droughts. The SPEI package in R was used to compute the index from the climate data.

Since the SPEI is usually used to characterise the duration, intensity, severity and frequency of extreme events, in this research it was used to identify events for exploring the models' predictions of wetland impacts during extreme events. Droughts and floods detected based on SPEI were further verified using historical records. Agricultural and meteorological droughts occur and are detected sooner than a hydrological drought. It generally takes several rainfall events to alleviate an area from its drought status and for the water storage infrastructure (dams for example) to return to levels between low and full supply. The SPEI only considers the difference between rainfall and PET and its value may no longer indicate severely dry conditions (implying a drought) when the drought is still in progress. To account for the initiation and continuation of droughts and its impact beyond months where the SPEI-12 < -1.5, a drought period was padded with 12 months prior to and after the index indicated dry conditions. Prior months were included in the event assessment period because the SPEI-12 index for a given month includes the rainfall in the 12 months preceding it.

To detect whether the extremity of dry and wet conditions have followed a monotonic trend during the assessment period, the Mann-Kendall test (Mann, 1945; Kendall *et al*., 1975; Ayugi *et al*., 2020) was applied to the series extreme events: SPEI-1 very wet months (i.e. SPEI > 1.5) and SPEI-12 very dry months (i.e. SPEI < -1.5), arranged chronologically, with the two sets being analysed separately. This non-parametric test determines the statistical significance of the alternate hypothesis: the data increasing or decreasing, linearly or non-linearly, over time. The significance of the trend was tested at 5 % ($p < 0.05$). When the results are statistically significant, the normalised test statistic (Z-score) expresses whether there is an increasing or decreasing trend. The test statistic (S-score) expresses whether the later values are more or less than the previous values. The Kendall test (tau) is analogous with the correlation coefficient in the regression analysis. The Sen slope estimator describes the rate (magnitude of change over time). The assessment was completed in python with the mannkendall package.

#### 4.2.6.3. Event analysis

For the selected drought and flood events in the assessment period, the modelled catchment streamflow and wetland net inflows, net outflows and change in storage were assessed. Catchment streamflow was assessed in terms of the hydrological impact from paired wetland versus no-wetland scenarios.

The change in wetland storage is a cumulative measure of the wetland water balance and reflects whether the wetland is predominantly releasing or storing water. A net release of water from the wetland storage (negative change in storage with inflows < outflows) indicated the wetland supplementing the catchment streamflow. A net storage of water inputs into the wetland (positive change in storage with inflows > outflows) indicated that the wetland was attenuating catchment streamflow. The wetland water balance flows were compared to the long-term annual averages (calculated for the water balance). The catchment streamflow was compared to the total hydrological impact for the assessment period.

For droughts periods, the total catchment streamflow and constituents (i.e. surface runoff, interflow and groundwater outflows) from the assessment period and individual drought periods were calculated.

## 4.3. Results

### 4.3.1. Model performance

#### 4.3.1.1. Streamflow

The ACRU models with the riparian zone (ACRU RZ model) wetland and wetland HRU (ACRU WL model) yielded satisfactory simulations of daily streamflow in terms of the distribution of flow values, but less so in matching the observed daily sequence. The total flows were simulated acceptably (Table 11). The total streamflow measured for the calibration period was 9.13 $Mm^3$. The RZ model overestimated the total streamflow as 11.31 $Mm^3$. The WL model underestimated the total streamflow as 8.89 $Mm^3$. Both models gave good estimates of the daily maximum streamflow: 2.57 $Mm^3$/day and 2.64 $Mm^3$/day for the observed value of 2.31 $Mm^3$/day. These peaks were slightly overestimated. The statistics show poor performance for simulating daily high flows with negative NSE values, largely due to mismatched timing of simulated peaks, with the models predicting flood peaks a day earlier than they were observed. It is important to note that the total flow in the calibration period was dominated by one large event, so the minor timing differences between the simulated and observed flows of this event are reflected in the statistics even though the magnitudes and peaks were successfully replicated.

*Table 11. Statistics for streamflow ($Mm^3$) modelled in the calibration period (October 2016 to September 2019) for the Upper K90A case study catchment*

| | WRSM | ACRU | | | | Observed data | |
|---|---|---|---|---|---|---|---|
| **Statistic** | CW<br>Month | RZ<br>Daily | RZ<br>Month | WL<br>Daily | WL<br>Month | Daily | Month |
| **NSE** | 0.68 | -0.28 | 0.74 | -0.32 | 0.75 | | |
| **NSE-logQ** | 0.72 | 0.30 | 0.40 | 0.17 | 0.23 | | |
| **KGE** | 0.73 | 0.37 | 0.50 | 0.40 | 0.57 | | |
| **$R^2$** | 0.77 | 0.18 | 0.96 | 0.17 | 0.95 | | |
| | | | | | | | |
| **Total Q** | 10.64 | 11.31 | 11.31 | 8.89 | 8.89 | 9.13 | 9.13 |
| **Min** | 0.00 | 0.00 | 0.02 | 0.00 | 0.00 | 0.00 | 0.00 |
| **Max** | 4.03 | 2.57 | 5.76 | 2.64 | 5.67 | 2.31 | 4.14 |
| **Average** | 0.30 | 0.01 | 0.31 | 0.01 | 0.25 | 0.01 | 0.25 |
| **Median** | 0.04 | 0.00 | 0.06 | 0.00 | 0.02 | 0.00 | 0.08 |
| **SD** | 0.80 | 0.08 | 0.98 | 0.08 | 0.97 | 0.07 | 0.68 |
| **CV** | 2.66 | 7.82 | 3.07 | 10.18 | 3.88 | 8.86 | 2.65 |

The ACRU models simulated similar hydrographs to one another in the calibration period at the daily time step. The ACRU models simulated daily low flows better than high flows with NSE (log Q) > NSE. The RZ model and WL model yielded NSE (log Q) values of 0.30 and 0.17, respectively, but was still unsatisfactory. Each model captured the minimum flow of 0 $Mm^3$/day as recorded in the streamflow records. The daily hydrographs show that the models did not simulate the receding limb (i.e. the transition from high to low flows) acceptably. Both models over simulated the volume and the time taken to return to recession flows (Figure 29). Statistically, the daily simulations of streamflow were moderately acceptable for simulating all flows, on average (KGE = 0.37 for the RZ model and KGE = 0.40 for the WL model). The linear regression of the simulated streamflow against the observed streamflow was also substandard with correlation coefficients less than 0.5.

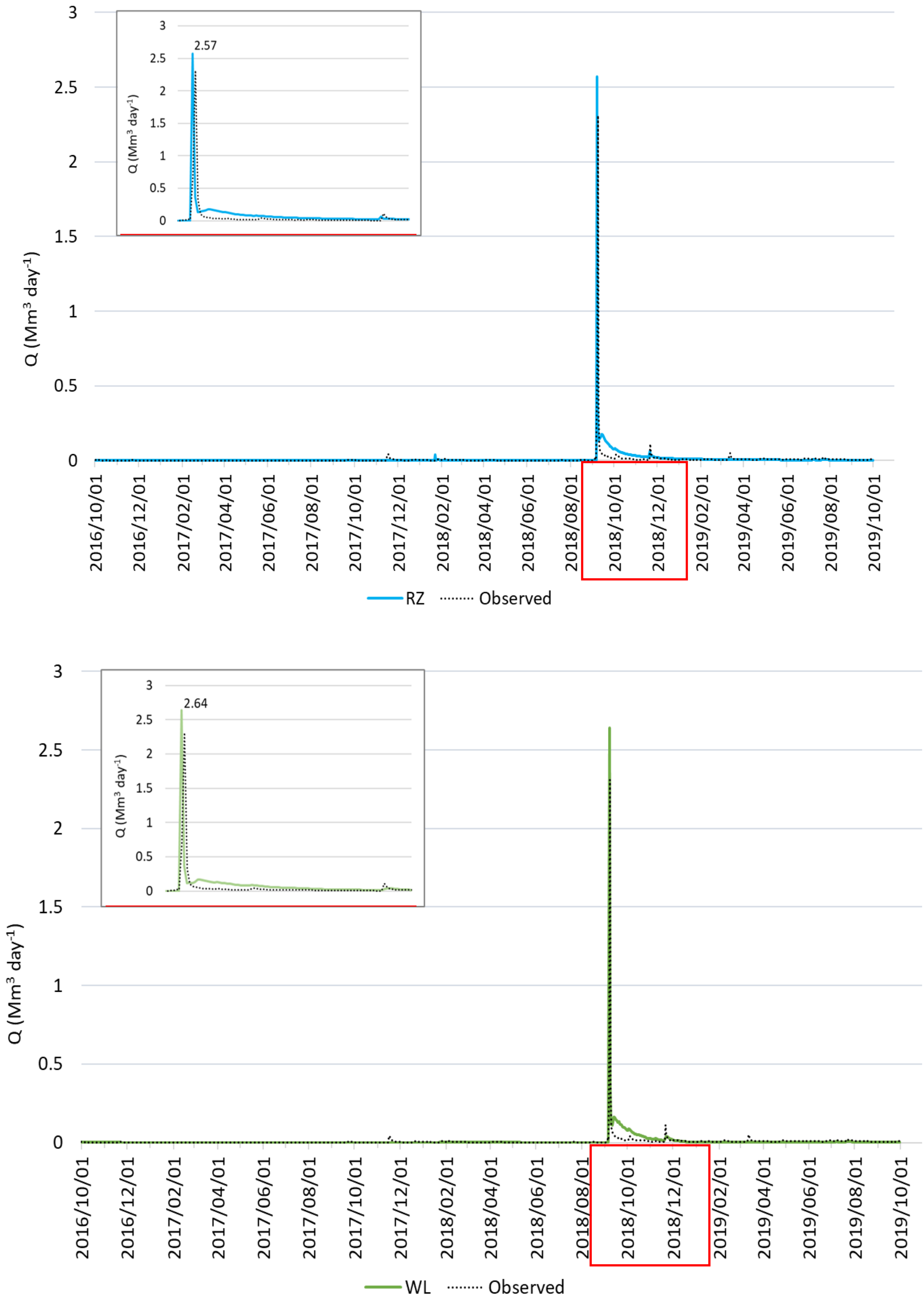



*Figure 29. Daily streamflow hydrograph for the ACRU models*

Good model performance on streamflow simulations was detected at the monthly time step. ACRUs daily streamflow upscaled to monthly volumes yielded better statistics (Table 11; Figure 30). The RZ and WL models simulated good monthly high flows (NSE = 0.74 and 0.75, respectively). The RZ model slightly improved the simulation of low flows with NSE (log Q) increasing to 0.40. The WL model still obtained poor performance for monthly low flows (NSE (log Q) = 0.23). In terms of monthly peak flows, the observed flows recorded a maximum of 4.14 $Mm^3$/month in October 2018. The RZ and WL models overestimated this total streamflow as 5.76 $Mm^3$/month and 5.67 $Mm^3$/month, respectively. The correlation coefficients from the linear regression significantly improved for both models to excellent status ($R^2$ = 0.96 for the RZ model and $R^2$ = 0.95 for the WL model). The observed mean streamflow of 0.25 $Mm^3$/month was simulated excellently by both ACRU models. The ACRU models continued to perform alike at the monthly time step.

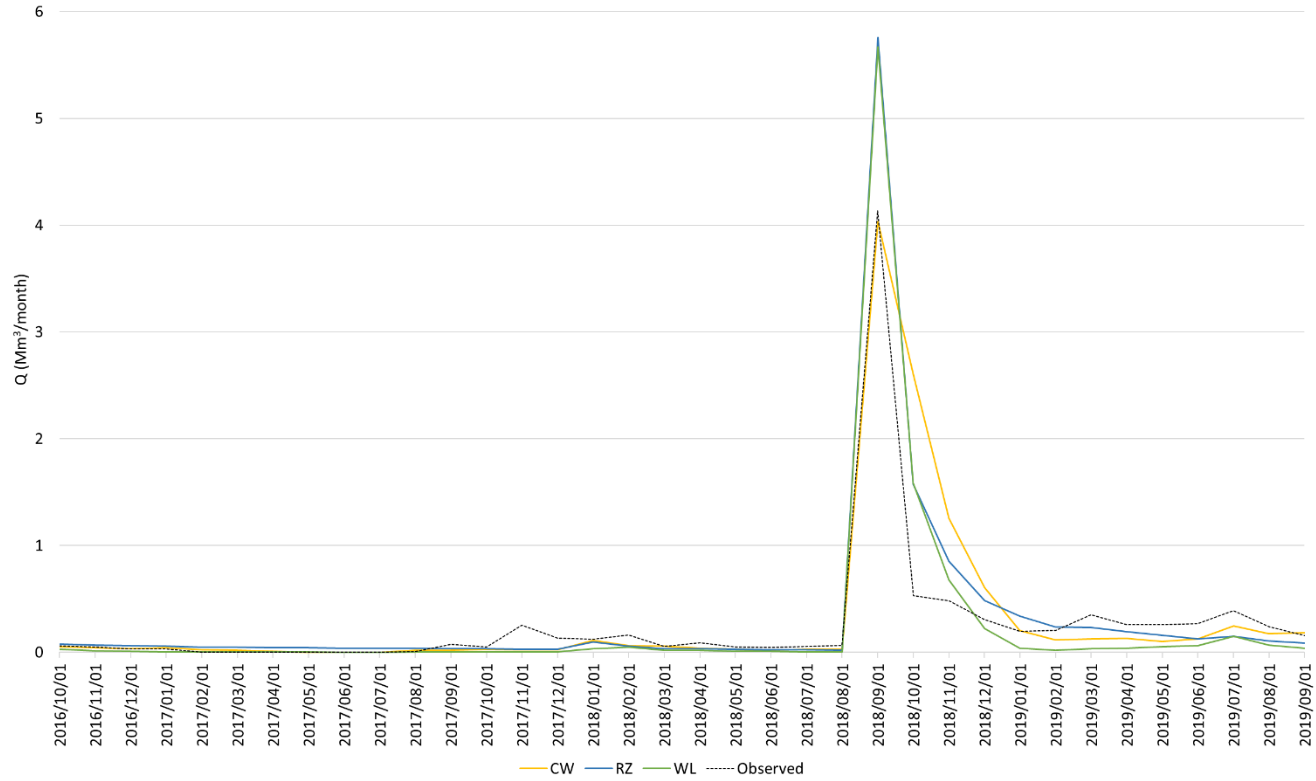



*Figure 30. Streamflow hydrograph in the calibration period for the models setup in WRSM-Pitman and ACRU tools*

The WRSM-Pitman model with the comprehensive wetland (CW wetland) simulated monthly streamflow acceptably for all sections of the hydrograph (Table 11). Monthly high, low and overall flows yielded statistics within the category of acceptable to moderately excellent model performance (NSE = 0.68, NSE (log Q) = 0.72 and KGE = 0.73). The correlation coefficient for the CW model was 0.77, indicating a good linear fit between the simulated and observed streamflow. The CW model over simulated the total streamflow for the calibration period (Q total = 10.64 $Mm^3$/month). This was less than the total streamflow over simulated from the RZ model. The CW model also captured the minimum flows of 0 $Mm^3$/month. An average streamflow of 0.30 $Mm^3$/month was simulated by the CW model, similar to the observed streamflow of 0.25 $Mm^3$/month. The hydrograph for monthly streamflow peaked at 4.03 $Mm^3$/month which was extremely close to the maximum in the observed flow records (Figure 30). The CW model, similar to the ACRU models, also over simulated the receding limb. However, the recession flows of the CW model matched the temporal distribution of the observed data set and was moderately under simulated. The recession flows for the ACRU models showed little temporal similarity with the observed flow records: the RZ model sharply declined to the end of the calibration period and the CW model flatlined to 0 $Mm^3$/month followed by a mildly under simulated response in August 2019.

#### 4.3.1.2. Actual evapotranspiration (AET)

Wetland AET was an auxiliary variable assessed to support the model performance evaluation based on streamflow. Simulated AET for the three wetlands was compared to external measures of AET (here after referred to as 'observed').

The WL wetland AET accurately and closely simulated the remotely-sensed FruitLook AET. FruitLook estimated a total of 1 520.30 mm evapotranspired from the palmiet wetland during the calibration period (Table 11). The WL wetland simulated 1 626.05 mm. Statistics derived from all monthly AET volumes (i.e. full data set of the calibration period) highlighted good model performance for high (NSE = 0.75), low (NSE (log Q) = 0.52) and all (KGE = 0.86) AET outflows from the wetland. The percentile distribution of monthly AET from the WL wetland almost perfectly simulated the FruitLook AET with the correlation coefficient increasing from 0.77 using the timeseries to 0.99 for the percentiles. Average and median monthly AET was also simulated acceptably by the WL wetland.

Wetland AET simulated by the CW and RZ models did not replicate the FruitLook AET. Table 12 shows that the RZ model under simulated the monthly FruitLook AET total as 946.08 mm, average as 39.42 mm and median as 26.67 mm. Poor model performance was indicated from the RZ wetland's full AET data set. On the other hand, the CW wetland over estimated AET compared to the FruitLook AET totals, average and median. CW model performance statistics for wetland AET were low. However, the statistics increased into acceptable ranges for the percentile distribution of monthly AET simulated by the CW and RZ wetland.

Considering the temporal distribution of wetland AET, the simulated AET responded differently before and after the 345.86 mm/month of rainfall in September 2018. This is illustrated in Figure 31. Prior to the rainfall events in that month, neither of the wetland models replicated the FruitLook AET. After this high rainfall month, the wetlands replicated the timing and volumes of AET estimated by the FruitLook algorithm. The CW and WL wetlands maintained good model performance in simulating AET after the rainfall events while the RZ wetland AET reduced sharply.

In terms of ground-based observations, all models under simulated AET measured from the scintillometer (Figure 31). Here, there was no correlation between the observed and simulated evapotranspiration timing or depths. Under simulated AET worsened in August and June 2019.

*Table 12. Statistics for evapotranspiration (mm) modelled in the calibration period (August 2017 to September 2019) for the Upper K90A case study catchment*

| **Statistic** | | **Full dataset** | | | **Percentiles[1]** | | |
|---|---|---|---|---|---|---|---|
| | | WRSM-Pitman | ACRU | ACRU | WRSM-Pitman | ACRU | ACRU |
| | | CW | RZ | WL | CW | RZ | WL |
| **NSE** | | -0.16 | -0.39 | 0.75 | 0.68 | 0.69 | 0.98 |
| **NSE-logQ** | | -0.19 | -1.21 | 0.52 | 0.60 | 0.57 | 0.93 |
| **KGE** | | 0.42 | 0.17 | 0.86 | 0.67 | 0.56 | 0.90 |
| **$R^2$** | | 0.29 | 0.10 | 0.77 | 0.87 | 1.00 | 0.99 |
| **Total** | *Observed* | | ***1 520.30*** | | | | |
| | Simulated | 2 054.80 | 946.08 | 1 626.05 | | | |
| **Average** | *Observed* | | ***63.35*** | | | | |
| | Simulated | 85.62 | 39.42 | 67.76 | | | |
| **Median** | *Observed* | | ***49.16*** | | | | |
| | Simulated | 77.75 | 26.67 | 51.62 | | | |

percentile range included the 10th, 20th, 30th, 40th, 50th (median), 60th, 70th, 80th and 90th percentile, including the minimum and maximum ET, see **Appendix 2** for data set

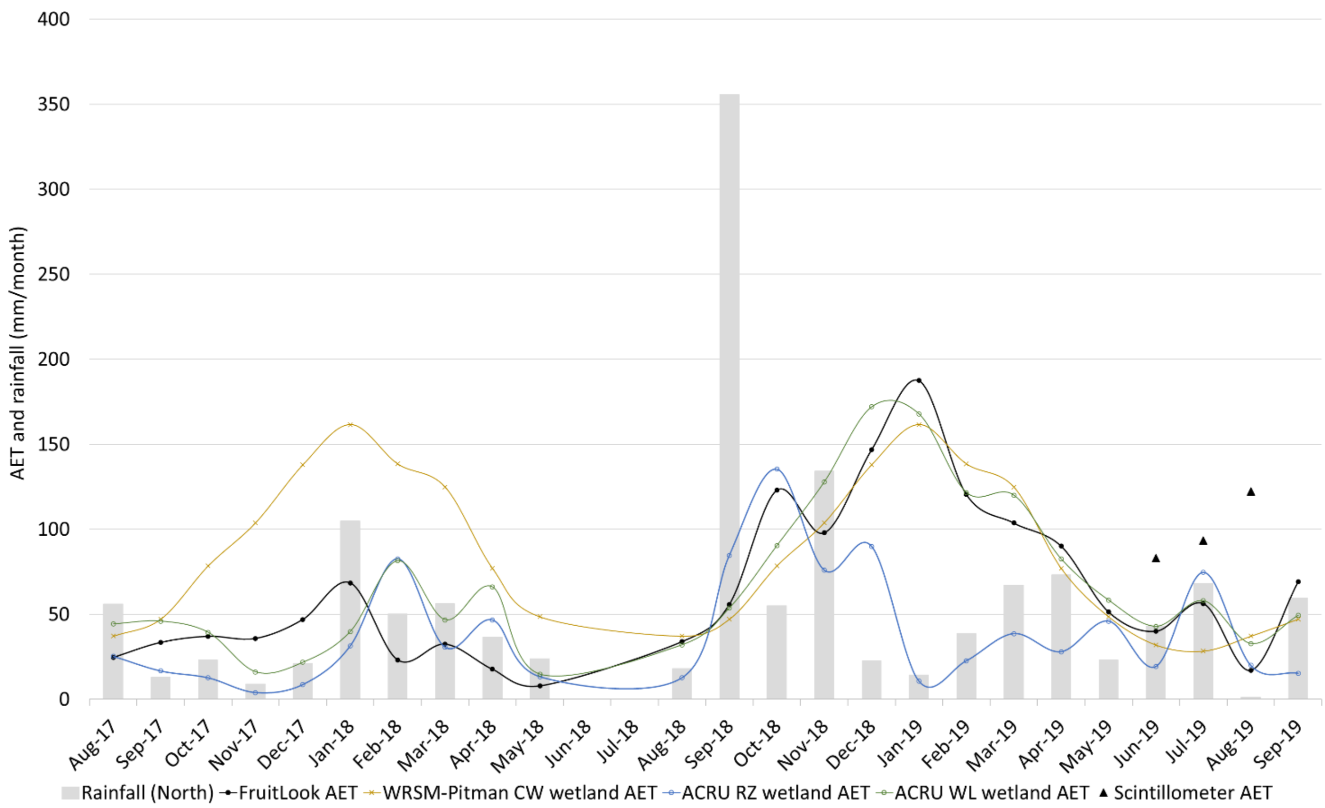



*Figure 31. Time series of observed, remotely-sensed (FruitLook, 2011 and updates) and modelled evapotranspiration from WRSM-Pitman and ACRU wetland models*

### 4.3.2. Modelled water balances

#### 4.3.2.1. Catchment

Figure 32 shows the predicted catchment water balances from each model for both the calibration (October 2016 to September 2019) and assessment (October 1989 to September 2020) periods. For both periods, the proportion of AET is similar within each model. The CW model does not differentiate between soil water evaporation and transpiration. The annual AET simulated fluctuates for each hydrological year. The results here refer to the annual average for the whole period. All models simulated larger annual average AET volumes in the assessment period. Comparing the AET volumes across models, the ACRU catchments simulated more AET (RZ wetland AET = 25.00 $Mm^3$ p.a. and 29.84 $Mm^3$ p.a., for the calibration and assessment period, respectively) than the WRSM-Pitman catchment (23.6 $Mm^3$ p.a. and 27.72 $Mm^3$ p.a. for the calibration and assessment period, respectively).

The models distinctly differ in their runoff compositions. The ACRU model's runoff consisted of mostly groundwater outflows ranging from 6.60 to 4.36 $Mm^3$ p.a. (Figure 32). Moving from the calibration period to the assessment period, the proportion of surface water and interflow outflows in runoff decreased for the RZ and WL model. The CW model simulated a large proportion of surface water and interflow in its runoff composition for both periods.

The annual rate of change for the storage units in the catchments were also distinctly different between models. The soil and groundwater storage of the CW model fluctuated in similar proportions. During the calibration period, the CW model simulated soil and groundwater storage declines of -0.21 $Mm^3$ p.a. and -0.32 $Mm^3$ p.a. In the assessment period, minor increases: the soil moisture storage increased by 0.04 $Mm^3$ p.a. and the groundwater storage increased by 0.02 $Mm^3$ p.a. Alternatively, the ACRU catchments predicted annual soil moisture storage decreases and groundwater storage increases. The groundwater storage increments of 0.01 $Mm^3$ p.a. were less than the soil moisture storage losses ranging from 3.81 $Mm^3$ p.a. to 6.00 $Mm^3$ p.a. on average from all hydrological years.

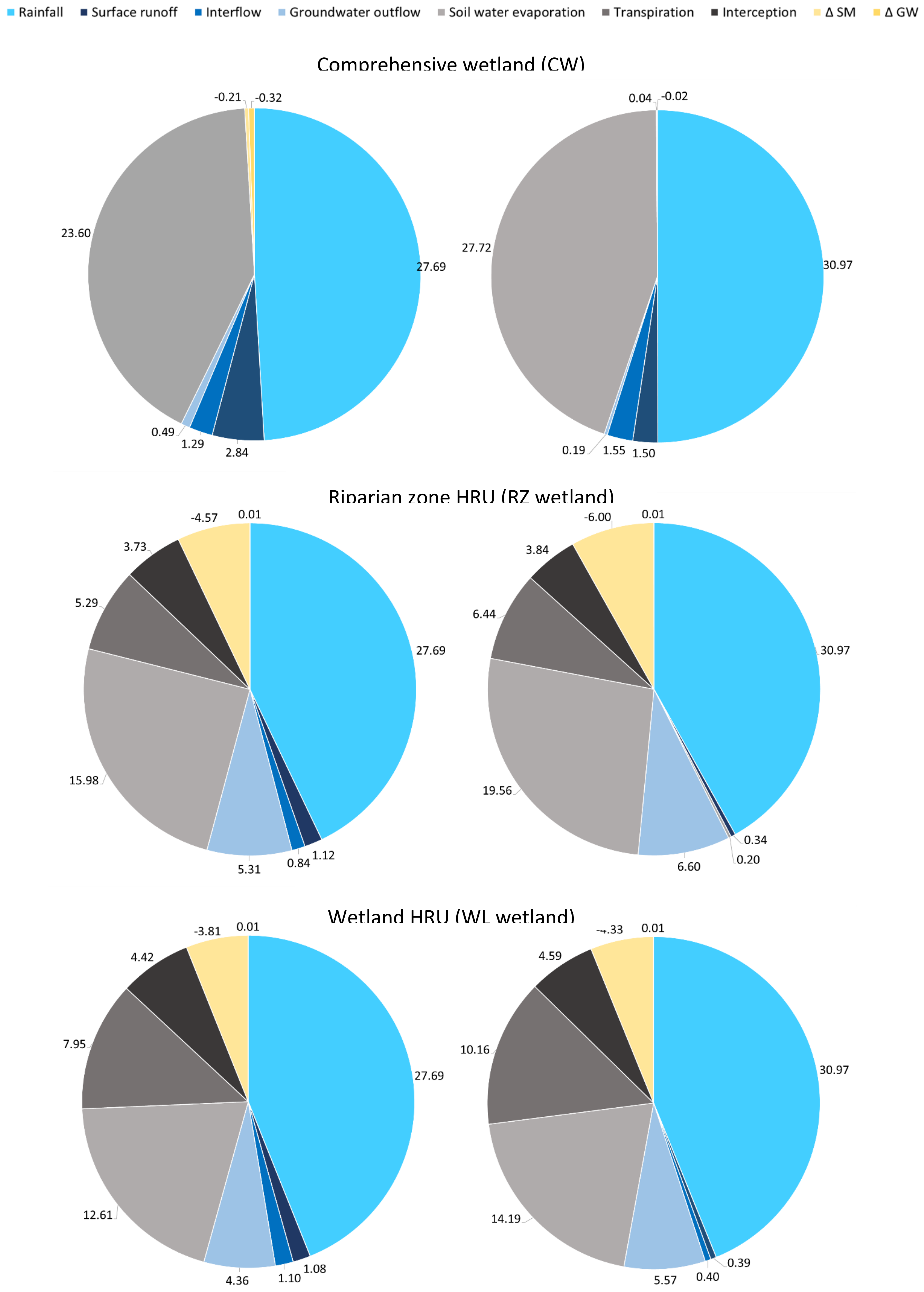



*Figure 32. Distribution of hydrological variables in the catchment water balance from the three models (with CW, RZ and WL wetlands) for the calibration period from October 2016 to September 2019 (left) and assessment period from October 1989 to September 2020 (right) presented as long-term annual averages in Mm$^3$ p.a.*

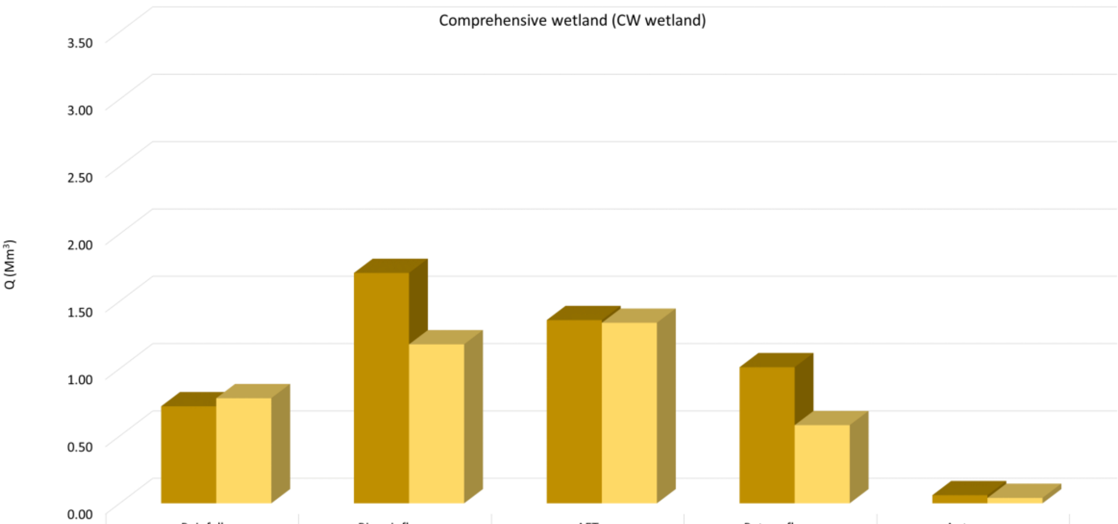
#### 4.3.2.2. Wetland

Figure 33 shows the predicted wetland water balances from each model for both the calibration (October 2016 to September 2019) and assessment (October 1989 to September 2020) periods. Flows in the wetland water balances are presented as annual averages from the hydrological years of the respective periods.

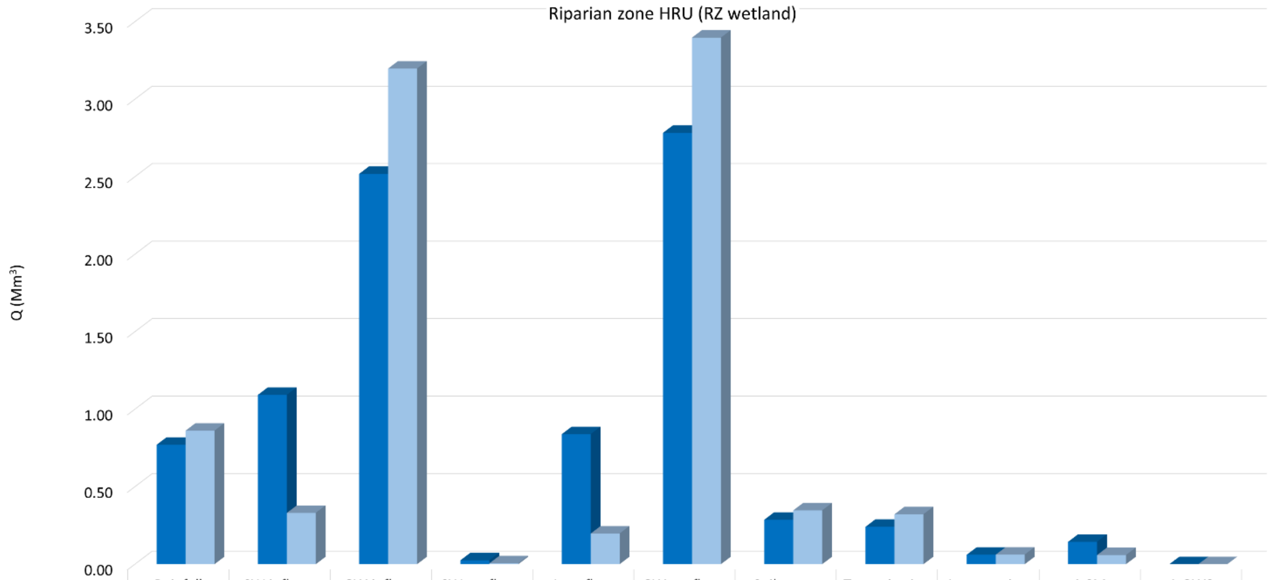
The CW wetland simulated more inflows than outflows with a net increase in storage in most years. The CW wetland received most of its inflows as river inflows in both simulation periods (1.71 $Mm^3$ p.a. and 1.18 $Mm^3$ p.a.). The average annual AET outflows from the CW wetland were similar in the calibration and assessment period. Concerning the outflow composition from the CW wetland, AET outflows were larger than the wetland's return flows to the downstream river.

Similar to the CW wetland, the RZ wetland predicted a net increase in the wetland storage in most hydrological years. Inflows into the RZ wetland were predominantly groundwater, followed by the surface water and rainfall inputs. In both simulation periods, rainfall was the lowest contribution to the inflow composition. The RZ wetland predicted most of its outflows as interflow and groundwater for the calibration and assessment period. An annual average of 0.06 $Mm^3$ p.a. was intercepted by the RZ wetland for both periods. The RZ wetland simulated similar volumes of evaporation and transpiration in each period. In the calibration period, 0.24 $Mm^3$ p.a. predicted to evaporate from the wetland's soil water and 0.24 $Mm^3$ p.a. transpired from the palmiet wetland. In the assessment period, the RZ wetland simulated 0.35 $Mm^3$ p.a. of soil water evaporation and 0.32 $Mm^3$ p.a. of transpiration. It was interesting to note observe that the groundwater inflows and outflows were similar for each simulated period and the largest volume of the inflow and outflow compositions.

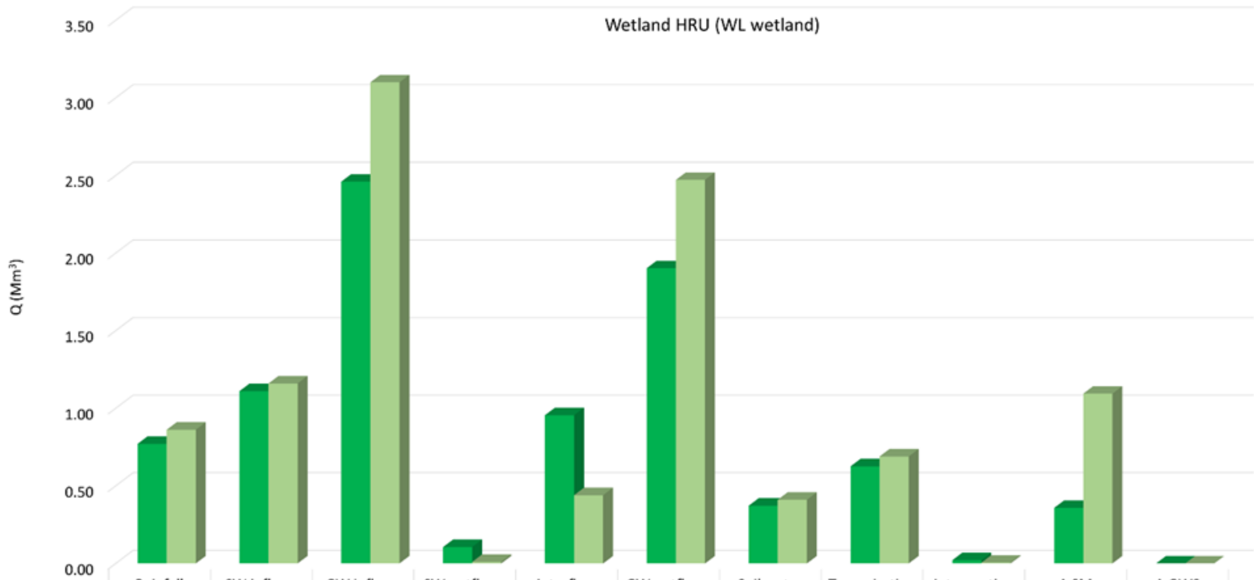
On average, the WL wetland simulated a net increase in the wetland storage for the calibration and assessment periods. Similar to the RZ wetland, the WL wetland inflows composed of groundwater, surface water and rainfall in order of contribution. In addition to this, groundwater formed the bulk composition of the wetland inflows and outflows. Interception was relatively similar between the calibration and assessment periods. However, different from the RZ wetland, the WL wetland simulated greater volumes of transpiration in both periods. The WL wetland modelled a long-term annual average of 0.69 $Mm^3$ p.a. transpired from the palmiet wetland in the calibration and 0.63 $Mm^3$ p.a. assessment period. The WL wetland simulated a larger annual average in the soil moisture and groundwater storage fluxes compared to the RZ wetland.




| | Rainfall | River inflows | AET | Return flows | Δ storage |
|---|---|---|---|---|---|
| Calibration period | 0.72 | 1.71 | 1.36 | 1.01 | 0.06 |
| Assessment period | 0.78 | 1.18 | 1.34 | 0.58 | 0.04 |




| | Rainfall | SW inflows | GW inflows | SW outflows | Interflow outflows | GW outflows | Soil water evaporation | Transpiration | Interception | Δ SM | Δ GWS |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Calibration period | 0.77 | 1.09 | 2.52 | 0.03 | 0.84 | 2.78 | 0.29 | 0.24 | 0.06 | 0.14 | 1.83E-06 |
| Assessment period | 0.86 | 0.33 | 3.20 | 0.01 | 0.20 | 3.40 | 0.35 | 0.32 | 0.06 | 0.06 | 8.37E-07 |




| | Rainfall | SW inflows | GW inflows | SW outflows | Interflow outflows | GW outflows | Soil water evaporation | Transpiration | Interception | Δ SM | Δ GWS |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Calibration period | 0.77 | 1.11 | 2.46 | 0.11 | 0.96 | 1.90 | 0.37 | 0.63 | 0.02 | 0.36 | 4.05E-07 |
| Assessment period | 0.86 | 1.16 | 3.10 | 0.01 | 0.44 | 2.47 | 0.41 | 0.69 | 0.01 | 1.09 | 6.49E-07 |

*Figure 33. Simulated long-term annual averages of wetland water balance fluxes for the three wetland models in the calibration (Oct 2016 – Sep2019) and assessment (Oct 1989 – Sep 2020) periods*

Comparing the three models, the wetland storage fluxes were similar, but the inflow and outflow compositions varied. All wetland water balances indicated that, on average, the annual wetland storages increased. The ACRU wetland's simulated larger wetland inflows and outflows than the WRSM-Pitman wetland. Regarding the outflow composition, AET was similar between the CW and WL wetland. The annual average wetland storage generally increased, especially in the WL wetland.

### 4.3.3. Hydrological impact of wetlands

#### 4.3.3.1. Total catchment streamflow in the calibration and assessment period

##### (i) CW wetland

Looking at the net effect modelled for the calibration period, the CW wetland attenuated streamflow. The CW model scenario without a wetland simulated 13.67 $Mm^3$. Adding the CW wetland decreased the total streamflow to 10.64 $Mm^3$ (Table 11). Table 13 shows that the CW wetland maintained the same overall regulatory role in the assessment period.

*Table 13. Long-term catchment streamflow totals and hydrological impact of wetland models in the assessment period*

| | Total Q ($Mm^3$) | | | | Influence | |
|---|---|---|---|---|---|---|
| ***Wetland model*** | *with wetland* | *without wetland* | ***absolute impact*** | ***relative Impact (%)*** | *general wetland impact on streamflow* | *predominant streamflow regulatory role* |
| **CW** | 72.15 | 99.70 | **-27.55** | -27.63 | decrease | attenuating |
| **RZ** | 111.60 | 109.36 | **2.24** | 2.05 | increase | supplementing |
| **WL** | 90.39 | 109.36 | **-18.96** | -17.35 | decrease | attenuating |

Flow duration curves for the paired scenarios applied over the calibration period, shown in **Figure 34**a, show that the CW wetland significantly attenuates high flows (viz. streamflow with a probability of exceedance less than 5 %). Medium streamflow volumes between probabilities of exceedance of 5 to 14 % were supplemented (i.e. the CW model simulated larger streamflow from the scenario with a wetland than the scenario without a wetland). Extremely low flows were attenuated by the CW wetland. Streamflow in the CW model reached 0 $Mm^3$/month sooner than the simulations without a wetland. The same trend, although more moderate, was observed for streamflow simulated during the assessment period (**Figure 35**a). However, medium flows were not predicted to be significantly supplemented by the wetland in general given the pattern of conditions experienced over the longer-term.

CW wetland filling was followed by spilling. **Figure 36**a presents the time series hydrograph for the CW model's paired scenarios. During the peak flows, the CW wetland attenuates flows. As streamflow recedes, the CW wetland supplements flows. Streamflow supplementation (i.e. the absolute and relative impact of the wetland or hydrological impact after the peak streamflow) was less than the extent of streamflow attenuation during the high flows. The same model behaviour of attenuated peak flows followed by streamflow supplementation repeats in the assessment period for the CW wetland as shown in **Figure 37**a.

**(ii) RZ wetland**

The RZ wetland simulated an overall net supplementing role on the catchment streamflow. The scenario without a wetland simulated 11.01 $Mm^3$ for the total streamflow in the calibration period, increasing to 11.31 $Mm^3$ for the scenario with the RZ wetland (Table 11). During the assessment period, the RZ wetland model also predicted that the wetland supplements streamflow on average (Table 13). **Figure 34**b and **Figure 35**b show that streamflow supplementation was not performed during extremely high or low flows but consistently, in small volumes, throughout the simulation period.

The time series of streamflow predictions for the RZ model shows long periods of consistent streamflow supplementation (**Figure 36**b). The initial very dry months of the calibration period showed no hydrological impact from the RZ wetland. However, supplementation began in January 2018 when conditions became slightly wetter. Peak flows were minimally attenuated in September 2018 after which the model returned to simulating streamflow supplementation. From the flood pulses identified from the rainfall records in section 4.2.6.2, the hydrological responses to events in 2006 and 2012 were used as additional high flow events for investigating the model behaviour. In these two events during the assessment period, the RZ wetland simulated the same hydrological impact of streamflow augmentation. **Figure 37**b illustrates the relatively consistent small supplementary influence of the RZ wetland on catchment streamflow. Compared to the other models, the hydrological impact predicted by the RZ wetland was relatively small and rarely exceeded 0.5.

### (iii) WL wetland

The WL wetland predominantly simulated an attenuative regulatory role on the catchment streamflow. Table 11 shows that in the calibration period, the total streamflow simulated for the scenario with a WL wetland decreased to 8.89 $Mm^3$. The attenuative role persisted in the assessment period. Table 13 indicates that the WL wetland had a hydrological impact similar to, but less than, the CW wetland and greater than the RZ wetland. The absolute and relative impact from the WL wetland in the assessment period was a reduction in streamflow by 18.96 $Mm^3$ and 17.35 %, respectively.

During the calibration period attenuation was primarily predicted to occur on medium and extremely low flows (**Figure 34**c). Attenuation on medium flows between probabilities of exceedance from 10 to 38 % was significantly noticeable. In the assessment period, the same model behaviour was observed. **Figure 35**c shows that the extent of attenuation decreased for streamflow less than 0.01 $Mm^3$.

The hydrograph for the WL model supports the finding that medium to extremely low streamflow volumes are attenuated. In **Figure 36**c, this hydrological impact was evident in the late receding to recession flows (i.e. the streamflow with a wetland was less than, or below, streamflow volumes predicted for the scenario without a wetland with a solid line). The 2006 hydrograph simulated in the assessment period shows the same predominate attenuation on catchment streamflow due to the WL wetland (**Figure 37**c). Streamflow attenuation was greater for non-peak streamflow sections of the hydrograph.

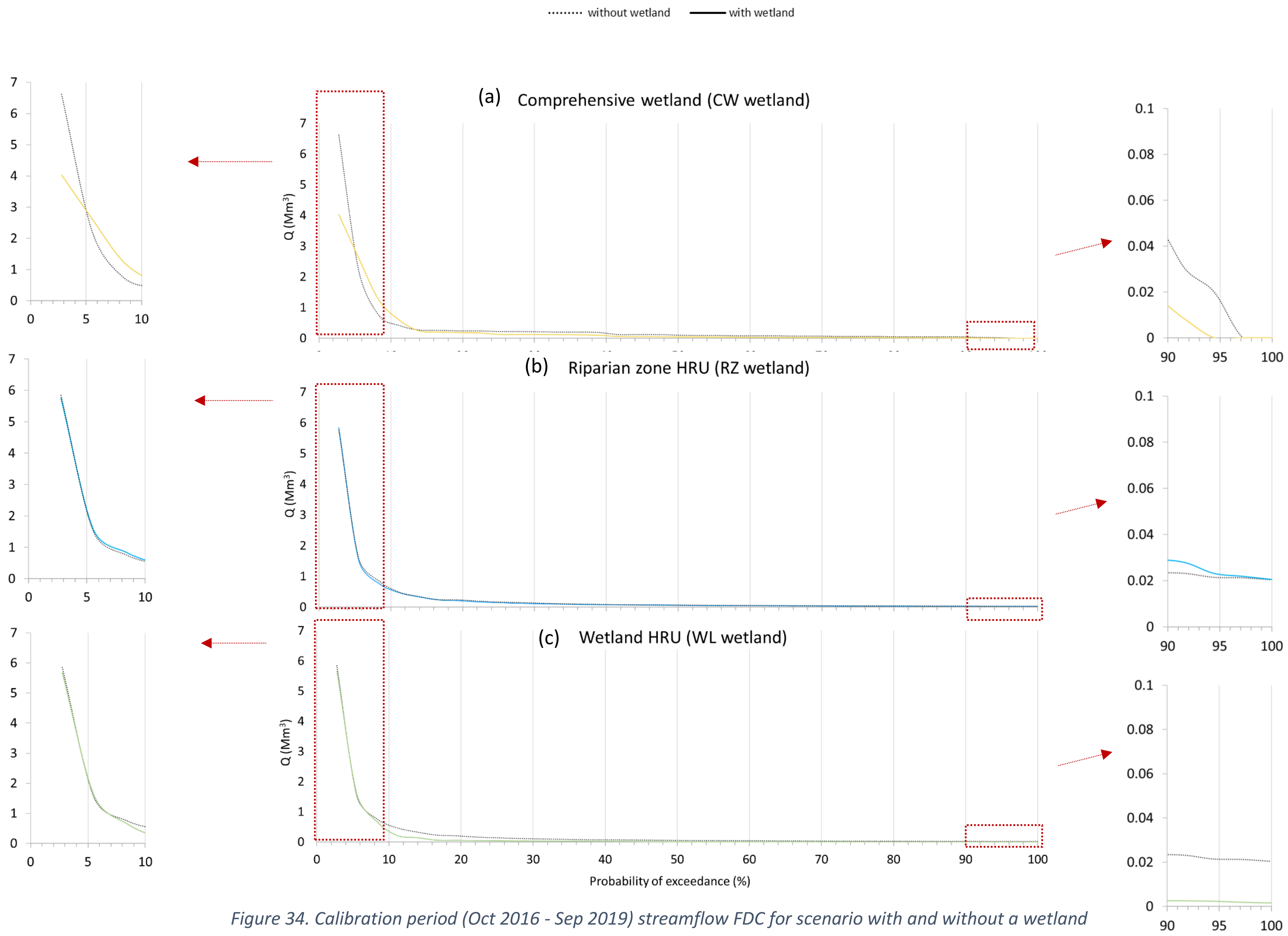



*Figure 34. Calibration period (Oct 2016 - Sep 2019) streamflow FDC for scenario with and without a wetland (from left to right: high, all and low flows)*

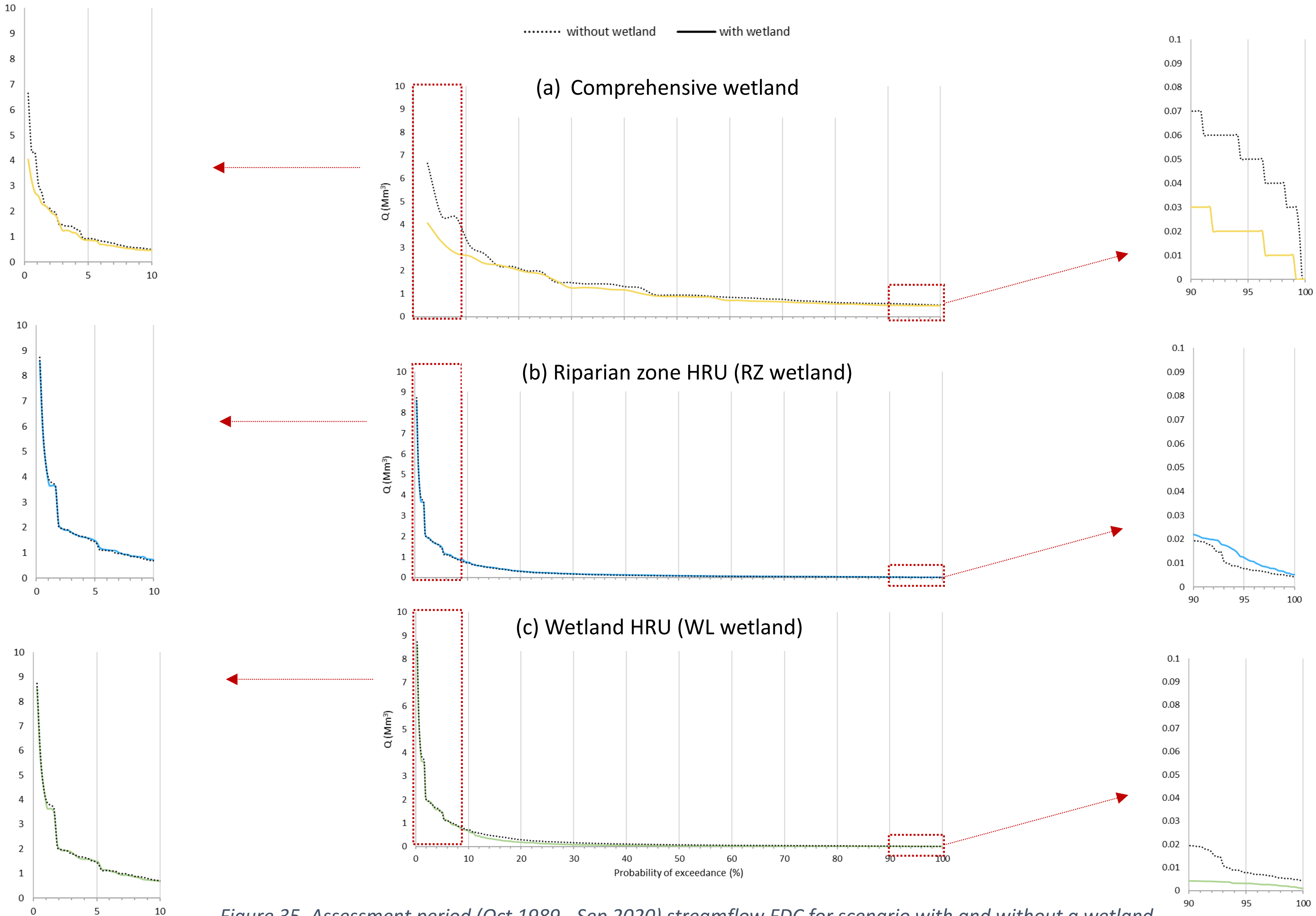



*Figure 35. Assessment period (Oct 1989 - Sep 2020) streamflow FDC for scenario with and without a wetland (from left to right: high, all and low flows)*

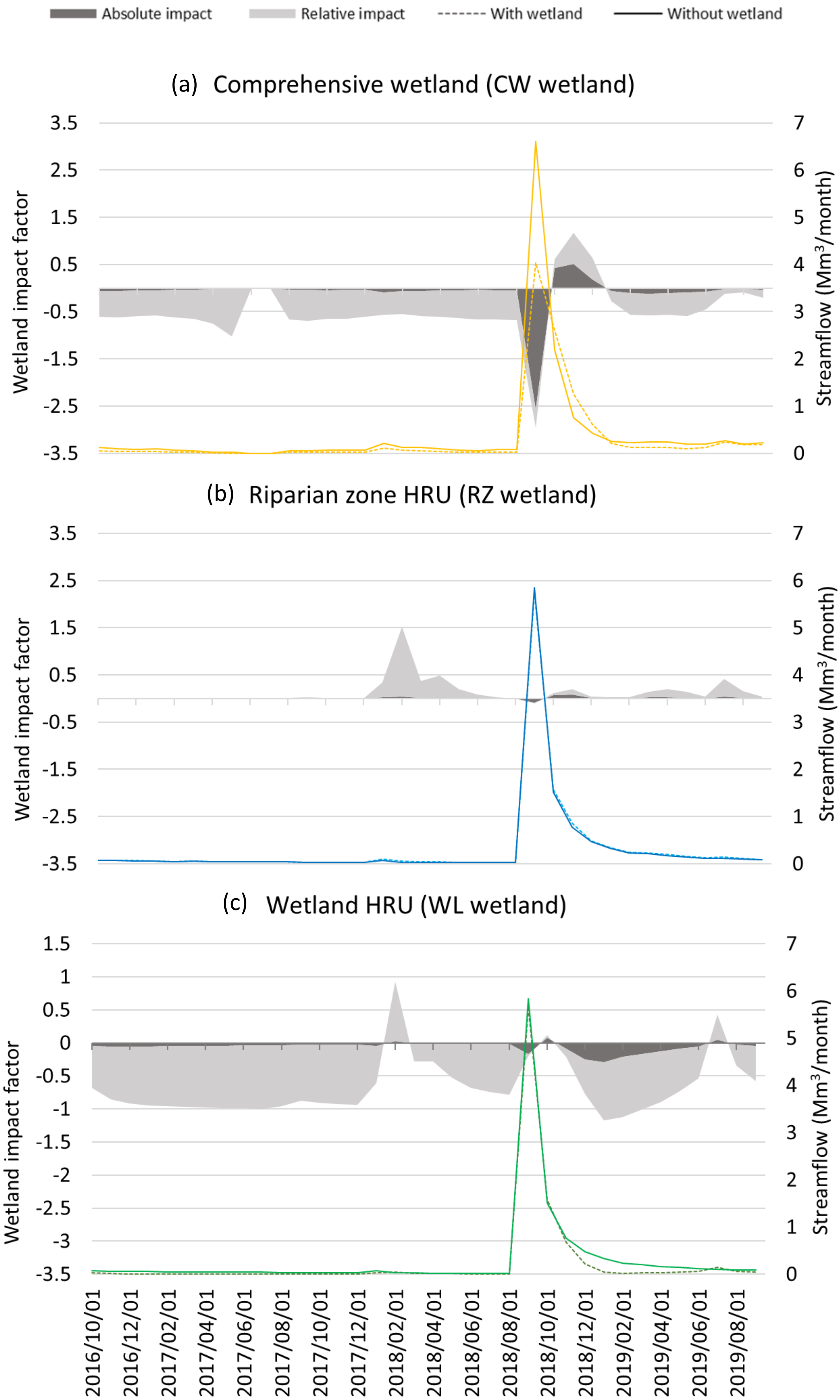



*Figure 36. Predicted hydrograph and hydrological impact of the Upper Kromme wetland on streamflow (calibration period) using three different models*

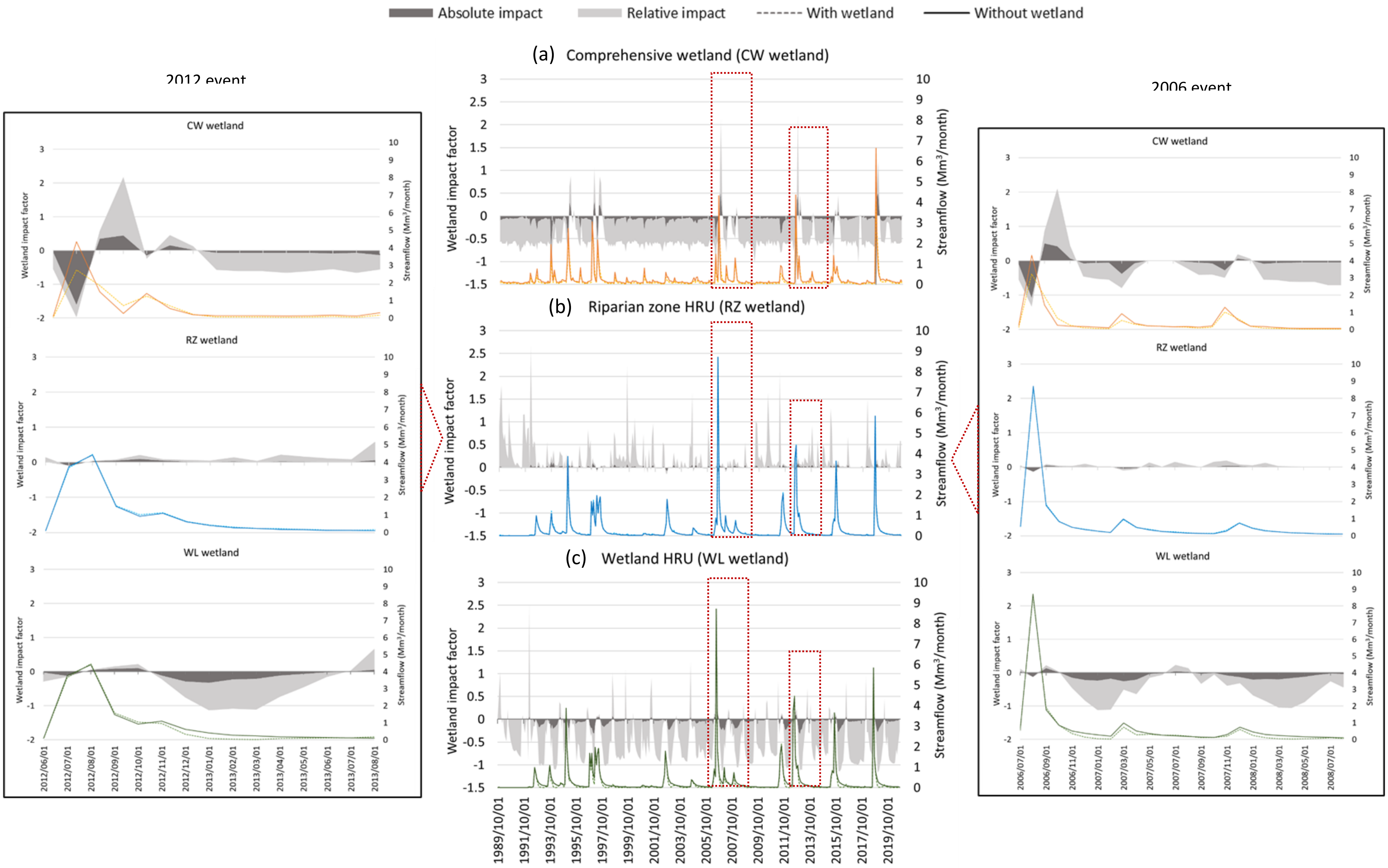



*Figure 37. Modelled relative and absolute hydrological impact of wetlands on streamflow during the assessment period and selected high flows*

#### 4.3.3.2. Wetland's regulatory role during extreme events

##### (i) Event detection

A statistically significant ($p = 0.03$) increase in the severity of droughts was detected in the assessment period (Table 14). Over this time period, floods were neither worsening nor becoming less severe. The MK test yielded statistically insignificant results ($p = 0.76$) for floods indicated by SPEI-1 ≥ 1.5.

*Table 14. Mann Kendall statistics on SPEI-1 ≥ 1.5 and SPEI-12 ≤ - 1.5 indicators for wet and dry conditions detecting the monotonic trend over time selected from the assessment period (October 1989 to September 2020)*

| | SPEI-1 ≥ 1.5 | SPEI-12 ≤ -1.5 |
|---|---|---|
| **Trend** | no trend | decreasing |
| **Ha** | FALSE | TRUE |
| **p, statistical significance (p <0.05)** | 0.76 | 0.03 |
| **Z, normalised test statistic** | -0.30 | -2.14 |
| **Tau, Kendall test** | -0.03 | -0.19 |
| **s, test statistic** | -8 | -30 |
| **Slope, Sen slope estimator** | -0.0059 | -0.0256 |

Table 15 presents the top five severe floods detected in the assessment period. Three of these floods were recorded in other literature sources. None of the floods occurred in the warm-up period and four events, where rainfall was concentrated to consecutive days, were retained for assessments in the extreme event analyses.

*Table 15. Flood events detected from the rainfall records (presented in order of peak rainfall depth)*

| Event | Date | Event rainfall (mm/day) | Peak rainfall (mm/day) | Duration (days) | SPEI-1 | Supporting literature and historical records |
|---|---|---|---|---|---|---|
| **1** | 2018/09/01 | 319.06 | 247.76 | 4 | 2.33 | EM-DAT, 2020 |
| **2** | 2012/07/01 | 275.63 | 202.13 | 6 | 2.16 | Pyle and Jacobs, 2015; Pyle, 2017 Mc Namara, 2018 EM-DAT, 2020 |
| **3** | 2006/08/01 | 272.02 | 201.74 | 5 | 2.36 | Ellery and Kotze, 2009 Mc Namara, 2018 EM-DAT, 2020 |
| **4** | 1996/11/01 | 240.39 | 172.69 | 7 | 2.22 | - |
| **5** | 1995/01/01 | 254.96* | 92.38 | 12 | 2.46 | - |

* 108.36 mm of rainfall from 1995/01/07 – 1995/01/11 followed by one day of no rainfall and then 146 mm of rainfall over seven days

Table 16 lists the top five droughts identified in the assessment period. All droughts were recorded in other literature sources. Two of the selected droughts were within the warm-up period and were excluded from the analyses of extreme events. The increasing severity of droughts highlighted in the MK test is evident in the SPEI-12 for the selected droughts. The minimum SPEI-12 was higher (i.e. less severe) for earlier drought periods (-1.71 for the 2008/09 drought). The later droughts showed the SPEI-12 decreased to minimum values of -2.30 in the 2017/18 hydrological year and -1.83 for the drought starting in 2019/20 (viz. droughts became more severe).

*Table 16. Severely dry conditions detected as SPEI-12 < -1.5 and the duration of the high index with records of regional droughts*

| ***Drought period[1]*** | ***Start*** | ***End*** | ***Duration (months)*** | ***SPEI-12 average*** | ***SPEI-12 min*** | ***SPEI-12 max*** | ***Supporting literature and historical records*** |
|---|---|---|---|---|---|---|---|
| **1** | Mar-17 | Feb-18 | 12 | -2.01 | -2.30 | -1.60 | Mahlela *et al.*, 2020 |
| **2** | Jun-20 | Sep-20 | 4 | -1.62 | -1.83 | -1.60 | Mahlela *et al.*, 2020 |
| **3** | Apr-83 | Jun-83 | 3 | -1.67 | -1.78 | -1.58 | Dube *et al.*, 2003 in Edossa *et al.*, 2014<br>Rouault and Richard, 2003<br>FAO, 2004 |
| **4** | Oct-84 | Nov-84 | 1 | -1.67 | -1.69 | -1.66 | Dube *et al.*, 2003 in Edossa *et al.*, 2014 |
| **5** | Feb-09 | Jun-09 | 4 | -1.62 | -1.71 | -1.50 | DoT, 2012 |

[1] drought period is not exhaustive to include only the months when SPEI-12 < - 1.5. It only indicates severely dry conditions (PET > rainfall) within an ongoing, longer drought period

### (ii) Floods

All wetland models simulated the attenuation of streamflow during floods. Table 17 presents the hydrological impact on catchment streamflow from paired scenarios and the monthly wetland fluxes for individual flood events.

According to the hydrological impact indicators, catchment streamflow was attenuated during flood events. The CW wetland modelled the maximal streamflow attenuation totalling 6.02 $Mm^3$ (total relative impact = 32.76 %). The WL wetland simulated the least streamflow attenuation (absolute impact = -0.42; relative impact = -2.48 %).

While attenuating flood waters, all simulated wetlands modelled an increase in the wetland storage. The ranking of the hydrological impact did not correlate to the same ranking in the wetland storage flux. For example, the WL wetland which had the lowest hydrological impact, simulated the largest increment in wetland storage of 9.44 $Mm^3$. The CW wetland modelled a similar storage increase of 7.27 $Mm^3$. The RZ wetland modelled the lowest wetland storage increase.

*Table 17.Hydrological impact on streamflow and wetland fluxes during flood events*

| | | Catchment streamflow (Mm³) | | | | | Dominant regulatory role | Wetland fluxes (Mm³) | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | With wetland | Without wetland | Absolute impact | Relative impact (%) | | | Inflows | Outflows | Δ Storage | Storage state |
| | Date | Total | Total | Total | Total | Average | | Total | Total | Total | |
| **CW** | 2018/09/01 | 4.06 | 6.64 | -2.58 | -38.86 | | attenuating | 3.14 | 0.22 | 2.93 | increasing |
| | 2012/07/01 | 2.75 | 4.36 | -1.61 | -36.93 | | attenuating | 2.17 | 0.21 | 1.96 | increasing |
| | 2006/08/01 | 3.24 | 4.32 | -1.08 | -25.00 | | attenuating | 2.19 | 0.76 | 1.43 | increasing |
| | 1996/11/01 | 2.31 | 3.06 | -0.75 | -24.51 | | attenuating | 1.63 | 0.67 | 0.96 | increasing |
| | ***Total*** | ***12.36*** | ***18.38*** | ***-6.02*** | | ***-32.76*** | | ***9.13*** | ***1.86*** | ***7.27*** | increasing |
| | | | | | | | | | | | |
| **RZ** | 2018/09/01 | 5.48 | 5.85 | -0.38 | -6.41 | | attenuating | 6.00 | 5.61 | 0.39 | increasing |
| | 2012/07/01 | 3.49 | 3.78 | -0.29 | -7.47 | | attenuating | 3.92 | 3.58 | 0.35 | increasing |
| | 2006/08/01 | 8.39 | 8.73 | -0.34 | -3.87 | | attenuating | 8.81 | 8.5 | 0.31 | increasing |
| | 1996/11/01 | 1.48 | 1.69 | -0.21 | -12.1 | | attenuating | 1.86 | 1.67 | 0.20 | increasing |
| | ***Total*** | ***18.84*** | ***20.05*** | ***-1.22*** | | ***-6.09*** | | ***20.59*** | ***19.36*** | ***1.23*** | increasing |
| | | | | | | | | | | | |
| **WL** | 2018/09/01 | 5.68 | 5.85 | -0.18 | -2.97 | | attenuating | 8.67 | 5.82 | 2.85 | increasing |
| | 2012/07/01 | 3.65 | 3.78 | -0.13 | -3.26 | | attenuating | 6.12 | 3.74 | 2.38 | increasing |
| | 2006/08/01 | 8.6 | 8.73 | -0.13 | -1.41 | | attenuating | 10.95 | 8.71 | 2.24 | increasing |
| | 1996/11/01 | 1.56 | 1.69 | -0.13 | -7.50 | | attenuating | 3.70 | 1.73 | 1.97 | increasing |
| | ***Total*** | ***16.59*** | ***17.00*** | ***-0.42*** | | ***-2.48*** | | ***29.44*** | ***20.00*** | ***9.44*** | increasing |

### (iii) Droughts

Table 18 presents the hydrological impact on catchment streamflow from paired scenarios and the monthly wetland fluxes for the individual drought periods detected in section 4.2.6.2, Table 16. Similar to the ranking of hydrological impacts in the analysis of flood events, the CW wetland simulated the largest hydrological impact on streamflow during droughts (absolute impact = -5.31; relative impact = -29.42 %), followed by the WL wetland (absolute impact = -3.97; relative impact = -27.92 %), and the RZ wetland (absolute impact = 0.67; relative impact 4.85 %).

The RZ wetland was the only model to predict streamflow supplementation during droughts. Catchment streamflow was supplemented while the wetland storage continued to increase. This model behaviour was observed for all drought periods.

The RZ wetland supplemented streamflow with groundwater outflows (Figure 38). During droughts, the main component of the wetland outflows was groundwater and interflow. This is similar to the long-term composition of the total runoff from the wetland.

The other models did not predict the expected role of unchannelled valley-bottom wetlands during droughts. The WL and CW wetlands simulated streamflow attenuation. Concerning the hydrological impact and wetland fluxes, the CW wetland storage modelled a net decrease in the wetland storage of 1.52 $Mm^3$. Increasing wetland storage was observed for one drought period during 2015/16 to 2018/19. Alternatively, the WL wetland storage consistently increased for all drought periods as streamflow was attenuated.

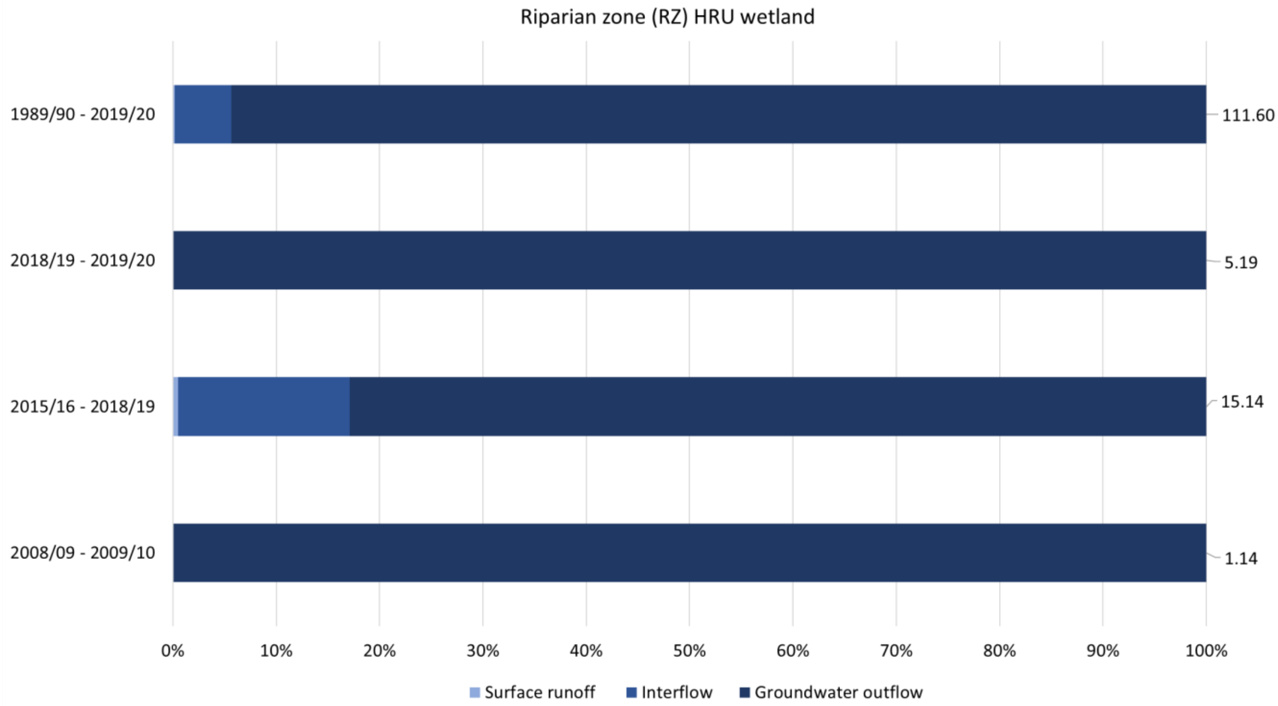
In terms of the runoff composition from the WL wetland during droughts, it was similar to the RZ wetland with large groundwater contributions to the runoff outflows (Figure 38). Unlike the RZ wetland, the WL modelled less total streamflow and a larger proportion of interflow (15 % in the assessment period and 28 % in 2015/16 drought).

The results indicate that long-term and drought-related hydrological impacts were similar. The CW wetland attenuated the total catchment streamflow for the assessment period (Table 13) and during droughts (Table 18). The WL wetland exhibited the same trend on catchment streamflow. On the other hand, the RZ wetland supplemented the total catchment streamflow for the assessment period (Table 13) and during droughts (Table 18).

*Table 18. Hydrological impact on streamflow and wetland fluxes during drought periods*

| | | **Catchment streamflow (Mm³)** | | | | | | **Wetland fluxes (Mm³)** | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | With wetland | Without wetland | Absolute impact | Relative impact % | | Dominant regulatory role | Inflows | Outflows | Δ Storage | Δ Storage model output | Storage state |
| | Hydrological year[1] | Total | Total | Total | Total | Average | | Total | Total | Total | | |
| **CW** | 2008/09 - 2009/10 | *1.16* | *2.56* | *-1.40* | *-54.69* | | attenuating | 1.87 | 3.11 | -1.25 | -1.38 | decreasing |
| | 2015/16 - 2018/19 | *10.05* | *13.03* | *-2.98* | *-22.87* | | attenuating | 6.98 | 6.79 | 0.19 | 0.17 | increasing |
| | 2018/19 - 2019/20 | *1.53* | *2.46* | *-0.93* | *-37.80* | | attenuating | 1.64 | 1.98 | -0.35 | -0.31 | decreasing |
| | ***Total*** | ***12.74*** | ***18.05*** | ***-5.31*** | | ***-29.42*** | ***attenuating*** | ***10.47*** | ***11.87*** | ***-1.40*** | ***-1.52*** | ***decreasing*** |
| **RZ** | 2008/09 - 2009/10 | 2.47 | 2.21 | 0.27 | 11.76 | | supplementing | 3.92 | 3.47 | 0.45 | | increasing |
| | 2015/16 - 2018/19 | 11.25 | 10.99 | 0.27 | 2.37 | | supplementing | 13.74 | 12.69 | 1.05 | | increasing |
| | 2018/19 - 2019/20 | 1.16 | 1.02 | 0.15 | 13.73 | | supplementing | 2.20 | 1.89 | 0.31 | | increasing |
| | ***Total*** | ***14.88*** | ***14.22*** | ***0.67*** | | ***4.85*** | ***supplementing*** | ***19.85*** | ***18.05*** | ***1.81*** | | ***increasing*** |
| **WL** | 2008/09 - 2009/10 | 0.87 | 2.21 | -1.35 | -60.63 | | attenuating | 4.13 | 2.93 | 1.21 | | increasing |
| | 2015/16 - 2018/19 | 8.90 | 10.99 | -2.09 | -19.02 | | attenuating | 17.04 | 12.02 | 5.02 | | increasing |
| | 2018/19 - 2019/20 | 0.50 | 1.02 | -0.53 | -50.98 | | attenuating | 1.91 | 1.64 | 0.27 | | increasing |
| | ***Total*** | ***10.27*** | ***14.22*** | ***-3.97*** | | ***-27.92*** | ***attenuating*** | ***23.08*** | ***16.59*** | ***6.50*** | | ***increasing*** |

[1] Hydrological year associated with the drought periods detected in Table 16 and padded with 12 months before and after the SPEI-12 < -1.5



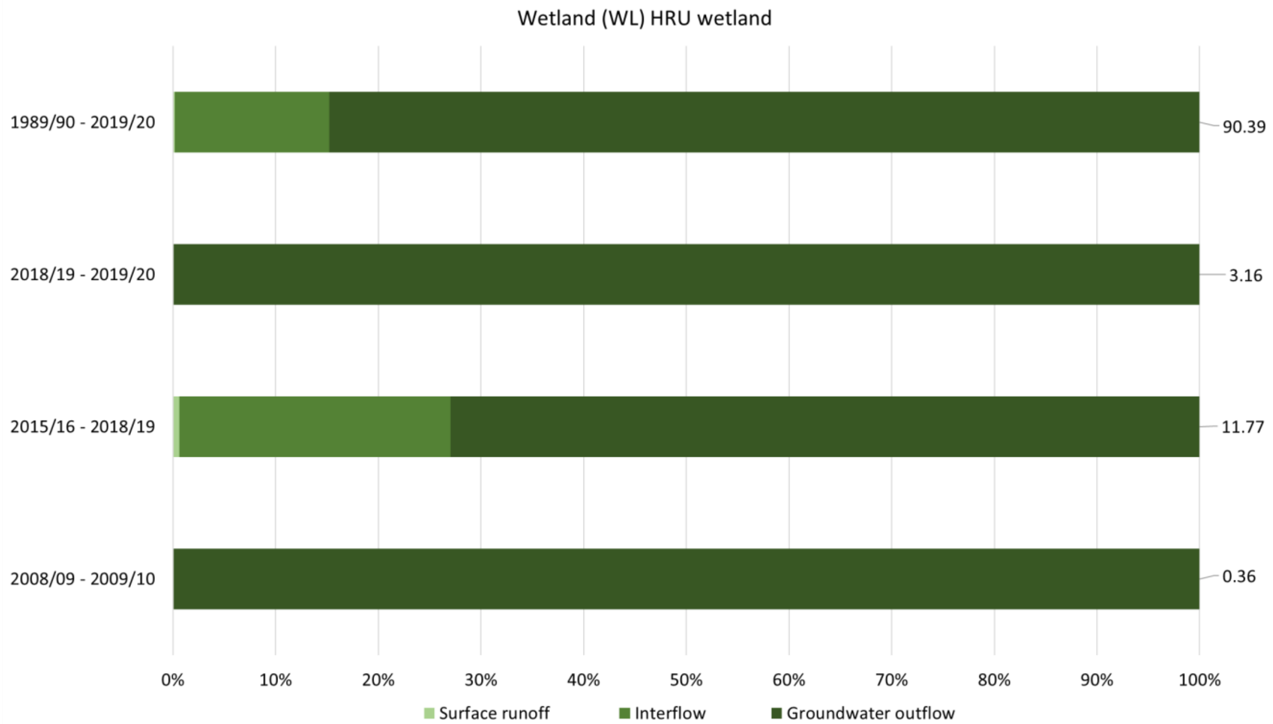




*Figure 38. Simulated runoff composition from the ACRU wetlands during the assessment period (October 1989 to September 2020) and three selected drought periods with the total streamflow (right) in Mm$^3$ for each drought period*

### (iv) Wetland storage

Figure 39 illustrates the cumulative wetland storage as monthly fluxes. Across all three wetland models, the long-term annual average change in wetland storage is similar (range: 0.01 – 0.03 $Mm^3$/month). The models are more responsive to floods which produced an increase in the wetland storage, especially the WL and CW wetland's. During droughts, the WL wetland storage simulated the greatest response, increasing by 0.08 $Mm^3$/month, on average. The magnitude of the wetland storage response was the same for the RZ and CW wetlands as 0.02 $Mm^3$/month. However, the RZ wetland storage increased while the CW wetland storage tended to decrease.

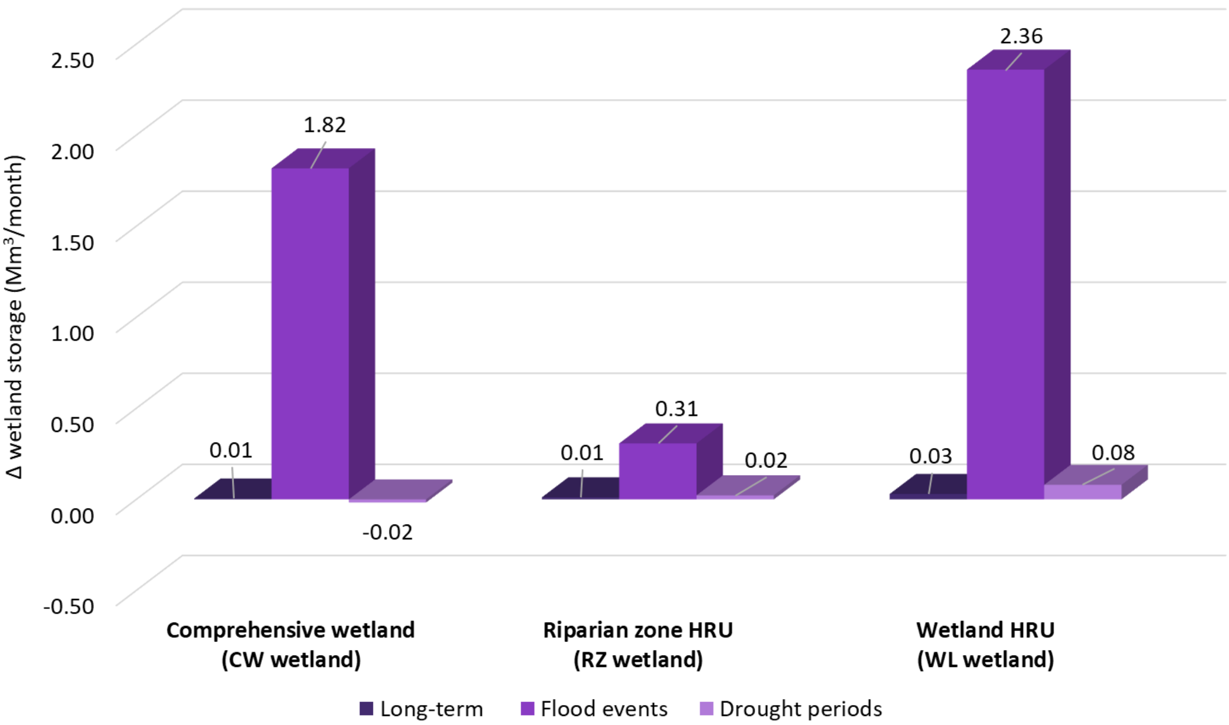



*Figure 39. Change in wetland storage for long-term average and extreme hydroclimatic events from ACRU and WRSM-Pitman's wetland models'*

## 4.4. Discussion

The quantitative analysis aimed to complete the estimation of wetland representation for the palmiet wetland. This was achieved by focusing on the hydrological functioning of the wetland as part of the catchment complex. Wetland representation has three aspects, namely, the coverage of wetland characteristics, processes and function (i.e. streamflow regulation). Modelled wetland functionality, which could not be addressed in the qualitative assessment, was addressed by quantitatively modelling the case study wetland within its catchment and climate context. The quantitative analysis and following discussion take into consideration that the wetland's predicted impact on catchment streamflow depends on the characteristics and processes defining the wetland model (i.e. the wetland model structure and parameters) and the simulated wetland's interaction with the climate and surrounding catchment.

The analyses in the results section addressed two research problems: first, identifying modelled streamflow regulation from the whole simulation period and in the subset of flood events and drought periods; and second, identifying the link between qualitative wetland representation in Chapter 3 and quantitative wetland representation deduced from modelling the catchment.

### 4.4.1. Summary of key findings

Variations in model performance are one of the main findings from this study. According to the convention that a credible model has acceptable model performance for variables in the catchment and wetland water balance (Evenson *et al*., 2018), the results suggest that ACRU's wetland HRU model (hereafter, the WL model) is superior to the other models. Good model performance was observed for the catchment and wetland water balance in the WL model. The WL model simulated acceptable model performance for streamflow outflows from the catchment (Table 11) and wetland AET at the monthly time step (Table 12). However, WRSM-Pitman's catchment configured with a comprehensive wetland (hereafter, the CW model) simulated acceptable model performance for streamflow while the wetland AET was poorly replicated. In addition to this, the catchment configuration in ACRU with a riparian zone HRU (hereafter, the RZ model) had substandard model performance with catchment streamflow over simulated and wetland AET under simulated.

Another key finding concerned the regulatory role of the wetland. The results indicate that the RZ model is superior to the other models as floods were attenuated (Table 17) and, during droughts, streamflow was supplemented (Table 18). Unchannelled valley-bottom wetlands are associated with streamflow attenuation and supplementation in response to high and low flows, respectively (Ollis *et al*., 2013; Mbona, 2016; Tanner *et al*., 2019). In terms of droughts and streamflow supplementation, the CW and WL models did not predict streamflow supplementation. Flood events were attenuated by all three models of the upper Kromme catchment.

Relating these findings to the wetland model suitability detected in the qualitative assessment of wetland representation (Table 4), high compatibility did not result in the simulation of both expected streamflow regulatory roles (e.g. attenuated floods and supplemented flows of droughts) or good model performance for streamflow and AET (i.e. both variables from the catchment and wetland water balance). By way of examples, the following relationships refer to Table 11 and Figure 30 for catchment streamflow model performance, Table 12 and Figure 31 for wetland AET model performance, Table 17 for flood events and Table 18 for drought periods. The highly compatible RZ model managed to simulate streamflow attenuation and supplementation but had poor model performance relative to the CW and WL models. On the other hand, the highly compatible WL model only simulated one streamflow regulation role expected of unchannelled valley-bottom wetlands despite having acceptable model performance for catchment streamflow and wetland AET. Alternatively, the moderately compatible CW model simulated one streamflow regulation role (i.e. flood attenuation) and surprisingly good model performance for one of the qualifying hydrological variables (i.e. streamflow). These results suggest that model specific processes and configurations, incorporated in the compatibility scores, could be associated with the likelihood of good performance for at least one water balance variable and the potential to simulate streamflow regulation. It may be that none of the models got the case study wetland correct and would probably need more data to determine which one is best. However, this is concerning considering that the Kromme wetland is one of the wetland sites with more data.

#### 4.4.1.1. Model performance

In terms of using a catchment and wetland water balance variable to validate a wetland model, the selected variables in this study were adequate indicators of model performance. In this case, the WL model was a superior approximation of the case study catchment and wetland. For the catchment water balance, streamflow has been widely used to reflect model performance. As such, the simulated streamflow can be relied on for hydrological and wetland model insights. For the wetland water balance, where wetland storage is the cumulative and fluctuating storage of the wetland water balance which has previously been used to identify credible models for catchments with wetlands (Evenson *et al*., 2018), there are examples of remotely-sensed AET effectively demonstrating a model's plausibility. In Ramatsabana *et al*. (2019), wetland AET was successfully estimated and validated against ground-based measurements with the FruitLook AET for the Krom catchment's palmiet wetlands. The study found that the remotely sensed AET product from FruitLook had the least under estimation of AET compared to other products. Furthermore, in Lee *et al*. (2021), remotely sensed AET gave the same indications on model performance as streamflow with changing model structures (i.e. where streamflow model performance improved with model structure enhancements, AET model performance also improved). To determine which models may be more credible, it would be reasonable to expect the likelihood of a model which is partly correct to have both a correct conceptual understanding the model setup and good model performance for the calibration period using additional data besides streamflow. Moreover, the modelled output indicates a complex interplay of variables which are often difficult to measure and rarely available, resulting in difficulties with creating and validating accurate wetland models.

In terms of model performance from ACRU, model processes and the flow gauging structure may have led to monthly simulations outperforming the daily model performance. During the model calibration, monthly statistics and the total streamflow were prioritised. The maximum flow and receding limb of the hydrograph were over simulated in the RZ and WL model at the daily time step (Figure 29). A parameter set which allowed the total, peak and receding limb to be simultaneously modelled well was not found. Other modelling studies have encountered this challenge and the likelihood for models to simulate one section of the hydrograph well at the expense of other segments (Lane *et al*., 2019). Timing errors in the daily streamflow were potentially carried over into the monthly hydrographs and statistics with relatively low NSE-logQ and KGE values for the monthly streamflow (Table 11). Additionally, the spatial scale of ACRU's wetland models is more suitable for the case study catchment and wetland size (hydrological response units within subcatchments) (Thornton-Dibb et al., 2010) as opposed to the CW module which was intended for larger, floodplain-like wetlands as noted in Maherry et al. (2017). The compatibility of the upper Kromme wetland's scale with the ACRU wetland model scale may have assisted the model performance improvements observed at the monthly time step.

Regarding the data quality, one potential reason for moderately acceptable model performance was the derivation of streamflow from rating curves using piezometer readings which are substandard to weirs and have a limited measurement footprint (WMO, 2012; McMillan *et al*., 2014; Dobriyal *et al*., 2017). Daily streamflow was probably less reliable than the monthly time step. In addition to this, the instrument was sub-optimally placed above the wetland bed which made it susceptible to periods where streamflow was below the instrument resulting in some extremely low flows being missed. Slightly further downstream, modelling the Kromme catchment and palmiet wetlands in SPATSIM yielded under simulated catchment streamflow for within the calibration period (Tanner *et al*., 2019). This may have been the result of limitations from the flow gauging structure or the model setup. Another study modelled the Kromme catchment up to the Kromme dam and reported good correlation between the dam inflow observations and the modelled streamflow (Rebelo, 2012; Rebelo *et al*., 2015). Although there are examples of successfully calibrating a model with piezometer data, this was completed in the context of a denser network of instrumentation with water levels in the topsoil peat and subsoil gravel compared with simulations of water levels from a fully-distributed model (House *et al*., 2016). This implies that the flow gauging structure may have made a difference in the streamflow observations and subsequent model performance.

Another potential reason for poor daily simulations may be related to the challenge of capturing complex, non-stationary processes. Spill-and-fill dynamics, wetland connectivity and the timing of these processes are difficult relationships to describe and encapsulate in models in a way that is applicable for most, or all, events. Hence, the ACRU models showed sensitivity to this challenge with poor model performance for daily simulations (Table 11). The challenge may have been further exacerbated by the uniqueness of the calibration period which was during a drought period and also contained the most severe flood on record (Table 15). Therefore, the model was calibrated during extreme conditions where soils may have been extremely, groundwater levels very low and water flow pathways may have been more disconnected. This could have also affected the lag of flows to, through and out of the wetland and was not captured in the available data or wetland model algorithms.

However, when ACRU's output was summarised as monthly values to be comparable with monthly output from WRSM-Pitman, model performance indicators improved but were still not excellent. Similarly, previous assessments have used longer time steps (in this case, monthly and annual volumes instead of daily simulations of streamflow, storage and annual average water balances) which has been associated with wetland models being too conceptual and simplistic to capture all hydrological processes occurring in the wetland at finer time scales (Dai *et al*., 2010; House *et al*., 2016; Muhammad *et al*., 2019). It is likely that the CW, RZ and WL models used in this research may be under the same limitations in their conceptual frameworks of wetland hydrology. Limitations and differences in the wetland models' processes, in particular spill-and-fill dynamics, are emphasised in the comparison of the ACRU models where the same subsurface properties and parameters lead to very different storage fluxes (for example, during floods, the RZ wetland gains less water than the WL model in Table 17).

Another factor which may have resulted in lower model performance are the conditions and length of the calibration period. A drought ensued during the calibration period (Table 11) and the 2018 flood pulse dominated the hydrological response of the period (Figure 29; Figure 30). These are extreme events with drastically different flow volumes and pathways which contribute to a complex and changing relationship between the wetland inflows and outflows that can be challenging to measure, understand and predict. The challenge encountered during the manual calibration was that parameters describing the rates of unsaturated and subsurface flow pathways at the daily time step could not be parameterised adequately without affecting the peak volume. It is possible that the models do not have the capacity to optimise both the timing of responses, peak volumes and total streamflow volumes. Concerning the record length, three years of streamflow data is relatively short to draw absolute conclusions on model performance. Fortunately, in the context of this research, a flood event and drought period were included in the calibration period. This suggests that the calibration period and setup is representative for the extremes assessed in the assessment period. This is similar to the findings of Bai *et al*. (2021) which suggested that the conditions of the model training should be similar to the simulation period to maintain similar model performance. Moreover, calibration and validation has been successfully completed for wetland catchments with shorter datasets (a period of 10 months each) for a more computationally and detailed model, MIKE SHE, compared to ACRU and WRSM-Pitman (House *et al*., 2016). Furthermore, all wetland models in this were relatively conceptual. A study by Bai *et al*. (2021) found that the length of calibration period does not affect model performance in validation period. Therefore, calibrating the case study wetland with three years of data was not a significant anomaly and was still somewhat reliable for model performance it was able to achieve and subsequent information the models suggested.

Moderate to low model performance can be associated with the link between information in the data records and the level of detail in the model's processes. For example, Makungu and Hughes (2021) reported that the wetland-river exchanges are not represented in the limited data available. In this case, the data records are unable to explain the movement of water into and through the wetland in a model that is either too simplistic or complicated relative to the actual processes occurring in the wetland. Furthermore, Wolski *et al*. (2006) reported that there needs to be a match between the scale of the model processes and data updating the wetland model of the Okavango Delta. Similarly, model performance has been shown to improve when the spatial discretisation of the model increased to represent the detailed data set (Muhammed *et al*., 2019). On the other hand, Hughes *et al*. (2013) state that model performance in data scarce areas and wetlands is higher in models with large time steps and spatial scales, as is the case for WRSM-Pitman, a lumped, monthly time step and model. This could explain why the CW model had better performance for the catchment streamflow. Comparatively, ACRU simulated the wetland and catchment units at the HRU level which has a finer level of detail which may not have been satisfied by the current flow records from piezometers which are substandard compared to gauging weir flows. However, considering that the water level was upscaled from hourly measurements to daily values, the daily mismatch of modelled flows may suggest that timings of the processes were not realistically represented in the wetland models of ACRU.

Additionally, some studies have reported variable and unsatisfactory model performance from paired scenarios (i.e. modelling the catchment with and without a wetland) explained by model processes as opposed to the temporal scale of the model. Modelling an unchannelled valley-bottom wetland (with no underlying, restrictive geology such that the wetland is in direct contact with groundwater) and seeps in SPATSIM, Maherry *et al*. (2017) suggested that reservoir fill-and-spill concept was inadequate for small wetlands. Here, model performance on streamflow statistics and the timing of flows was the result of insufficient process coverage in the model. Modelling floodplain wetlands in ACRU and SPATSIM, Mandlazi (2017) associated satisfactory model performance for the catchment streamflow with insufficient groundwater dynamics in the data and models together with limitations in the observed flow data in their case study modelling. As part of the recommendations for future research, Maherry *et al*. (2017) proposed modelling wetlands at the daily time step as potential improvement to monthly time step simulations. However, simulations in Mandlazi (2017) and this research indicate that daily time step modelling does not necessarily improve model performance. This suggests that the concepts underlying the wetland models are not yet sufficiently describing the wetland processes. Examples of processes not yet captured in the daily models may include soil drainage and rewetting hysteresis or rates at which surface and subsurface flow pathways connect and disconnect with the wetland during and after a rainfall event. However, for modelling riparian wetlands with no seepage, the output and concept in ACRU wetlands are relatively acceptable. A verification process by Gray (2011) found that the wetland responses in ACRU are hydrologically sound for perched, riparian wetlands. This suggests that the wetland model was suitable for the riparian case study wetland with no seepage to an underlying aquifer. This may have resulted in streamflow model performance in the RZ and WL models which was not excellent, but acceptable.

Another challenge that may have affected the model performance of catchment streamflow was keeping the catchment and wetland configurations consistent across the selected models. Few model properties and parameters were transferable between WRSM-Pitman and ACRU. This was specifically true where either wetland model had processes, properties or parameters absent in the other tool. For example, WRSM-Pitman's division of upstream river flows into through flows and wetland inflows or ACRU's parameter separating incoming runoff into same day and delayed wetland outflows. The models were kept similar as far as possible then calibrated according to the modelling tool's recommendations. This was more difficult to maintain for the wetland models, especially considering that the ACRU and Pitman models have different conceptual perspectives and algorithms. WRSM-Pitman's CW wetland depended on river inflow thresholds and storage while ACRU's wetland models were more dependent on the soil saturation and infiltration rates.

Restricted model performance for the catchment streamflow, ranging from acceptable to good, could also be attributed to static, simple or average wetland or catchment properties and processes in the models. At site scale, wetland and subsurface properties can vary, even in riparian valley-bottom wetlands as small as 10 ha (0.1 $km^2$) (House *et al*., 2016), however, fixed values representing the whole wetland were used in the models. Considering that the case study wetland area is larger than this (1.5 $km^2$), it is highly likely that variability in the wetland and subsurface properties were not accounted for and may have limited the model performance.

Similarly, uniform soil depth was assumed for the whole wetland despite evidence for the wetland's soil depth varying. In-situ observations reported deeper depths at the wetland inflow and shallower depths towards the wetland outlet (Pulley *et al*., 2018; Lagasse, 2017). However, the average wetland depth, adjusted by in-situ measurements of porosity, was used to calculate the wetland storage. It is possible that this estimate of wetland storage may have over or underestimated the real wetland storage. Uncertainty in this value may have affected the models' calculations of water that can be retained in the wetland and subsequent streamflow outputs resulting in variable model performance.

Furthermore, in terms of wetland storage properties representing the physical wetland's substrate, none of the models were able to, nor configured for, changing soil properties. The degradation of peat leads to changes in the physical properties of the wetland's soil profile, or storage, which affects the amount of water that the wetland can retain or that passes through the wetland. Therefore, modelling peats is challenging because of these changing properties and variable water flow through the material. Changing peat properties has been noted as a cause for reduced model performance when modelling a riparian valley-bottom wetland with peat in House *et al*. (2016). Similarly, with the effects of soil deformation and changing properties of the peatland (i.e. wetland's soil/storage profile) for the case study wetland not incorporated into the models or modelling, this may have reduced the model performance.

Providing more evidence for the potential impact of averaged inputs in models on model performance, a previous study modelling coastal forested wetlands, Dai *et al*. (2010) linked average characterisations to good model performance for average conditions only. Similarly, this may explain why the three models (namely, the CW, RZ and WL models) achieved moderately acceptable model performance for streamflow instead of excellent results for any flow segment of the hydrograph (e.g. high, medium or low flows).

In addition to this, neither surface water-groundwater interactions nor groundwater flows were accounted for in any of the models with great detail. In terms of groundwater representation in the respective models, the groundwater storage was parameterised simply with outflows based on the current volume in ACRU's wetlands while there was no groundwater storage in the WRSM-Pitman wetland. According to model development for the Okavango Delta (Wolski *et al*., 2006), groundwater storage and flow based on the volume in the surface groundwater reservoirs is too simple compared to the explicit representation of the storage and algorithms using water levels and physically-measured properties (e.g. hydraulic conductivity and transmissivity) to estimate flows from the groundwater storage of the wetland.

Furthermore, modelled water flows in and out of the wetland were in one direction. In all models, water moved into the wetland from the surrounding catchment and out of the wetland to the catchment outlet. No water movement from the wetland to surrounding alluvial aquifers or land uses was represented (viz. the potential for water to move in multiple directions). There may have been instances where water movement in variable directions would be plausible for the case study wetland due to hydraulic gradient between the wetland storage and alluvial aquifers and the presence of peat. Even though ACRU and WRSM-Pitman model structures are not designed to explicitly capture physically-based processes at point scales, the anisotropic movement of water in peatlands could not be captured in MIKE SHE either which has a finer spatial resolution and physically-based algorithms (House *et al*., 2016). As with this example, in the current modelling exercise, simple and unidirectional water movement could be another cause of less than excellent model performance.

Concerning the model type and scale, model performance was better in the modelling tool which is more physically-based and distributed. ACRU is a conceptual, physically-based and semi-distributed modelling tool which was configured in this way for the case study catchment. Alternatively, WRSM-Pitman is a conceptual and lumped model. With respect to the highest qualifying standards of model performance using a catchment and water balance variable (i.e. streamflow and wetland AET), this research observed good model performance in WL model configured in ACRU. Similar findings of good model performance in more process-based, distributed and complex models have been found when modelling catchments without wetlands (Okiria *et al*., 2022) and catchments with wetlands (Dai *et al*., 2010; House *et al*., 2016; Wolski *et al*, 2006). Altogether, these studies attributed improvements in model performance to more physical knowledge and measurement-based parameters incorporated in the model structure compared to calibrating parameters which are mostly or entirely conceptual, the inclusion of spatial complexity in the system and detection of hydrological dynamics regulating the wetland storage. This suggests that groundwater in channel flow of the main river was a viable model for the upper Kromme, palmiet wetland. Additionally, the model performance suggests that hydrological processes in the catchment and wetland configured in ACRU are more credible representations of the case study hydrology.

Using the traditional qualifying criteria of streamflow only, all models had acceptable model performance. In the case of the more conceptual model, WRSM-Pitman, one potential reason for good model performance for catchment streamflow was found in Dai *et al*. (2010) where lumped models were able to perform reasonably well at the monthly scale and for average conditions. Good model performance for all sections of the hydrograph may have also been an extension of a match between the complexity and quality of streamflow records with the model structure. This suggests that WRSM-Pitman may be capable of modelling small wetlands which are currently not adequately represented in Pitman-SPATSIM (Maherry *et al*., 2017). Concerning ACRU's wetland models, including subsurface inflows into the wetland and having a compatible spatial scale with the case study wetland improved hydrological realism. Additionally, streamflow total's and maximum values for ACRU wetlands were acceptably simulated. Therefore, these findings demonstrate and confirm the link between physically-based models and distributed models leading to better model performance and the exception, or conditions, where conceptual models are credible.

Providing another potential reason for the CW model's good streamflow performance for all flow volumes, it is possible that this model captured the wetland-river relationships at the monthly scale and to the same level of detail available in the observed data sets. Makungu and Hughes (2021) suggest hydraulic conductivity as the fundamental principle of modelling floodplain-river exchanges. Since the CW model simulated all sections of the hydrograph acceptably, this implies that the parameters describing water movement between the wetland and river are sufficient estimators of the hydraulic gradient between the wetland and river. A model which simulates all flow segments in a hydrograph is very appealing because most models are good simulators of parts of the hydrograph with high flows taking precedence in the calibration process (de Boer-Euser *et al*., 2017), or a few processes in the hydrological cycle (Seiller *et al*., 2012; Lane *et al*. 2019). This was the case for the ACRU models where peaks and totals were modelled well but the receding limb of the hydrograph, volume of low flows and timing of the events were sometimes variable compared to the observed streamflow. Furthermore, simpler models, in terms of spatial aggregation and algorithm complexity, have been noted as suitable for data scarce catchments (Hughes *et al.,* 2013; Makungu and Hughes, 2021).

#### 4.4.1.2. Streamflow regulation

Taking into account the model type and input, the wetland models in more physically-based, distributed model had better model performance (viz. from the WL model) and predicted streamflow regulation roles associated with an unchannelled valley-bottom wetland (viz. from the RZ wetland). Firstly, where Dai *et al*. (2010) found that distributed models give more accurate predictions of streamflow water table depths in varying climatic conditions, the same was found in this research with the RZ model from the semi-distributed modelling tool, ACRU, simulated streamflow attenuation and supplementation during floods and droughts, respectively. Secondly, where Dai *et al*. (2010) proposed that complex (i.e. more physically-based and distributed) and simpler (i.e. more conceptual and lumped) models generate the same hydrological responses under average and long-term conditions, this was not the case for the models investigated. The reasoning for this logic was that the input and setup in the simpler models represent average conditions while the complex models yield more accurate predictions of streamflow, wetland storage and water table depths in varying conditions since the model configuration is more differentiated. The differentiation allows for hydrological processes in the model to capture spatial heterogeneity in the catchment and the variations in outflows from the catchment (Wolski *et al*., 2006; Dai *et al*., 2010). However, the hydrological impact from the CW wetland (relatively simple model) was similar to the response simulated by the WL model (relatively complex model) (Table 9). During floods, the models simulated similar attenuative response, suggesting no confirmation of the link between model complexity with model performance and streamflow regulation. During droughts, the complex and simple models simulated net streamflow attenuation and mostly wetland storage increases, contrary to expectations from classification literature. One model, the RZ wetland simulated streamflow supplementation during droughts. This model was associated with the more complex model and confirms the idea that more physically-based, semi-distributed models but also had realistic subsurface storage and flow pathways, improving the wetland model's ability to simulate streamflow regulation for the case study wetland. Conclusively, these results suggest that the link between model type and input with the simulation of streamflow regulation expected from a wetland is context specific, variable and currently only held true during droughts.

In addition to the wetland typology, streamflow regulation is a reasonable expectation based on the case study wetland's location. According to the classification of wetlands as hydrogeomorphic units, the case study wetland is an unchannelled valley-bottom wetland (Ollis *et al*., 2013). This wetland type is associated with moderately attenuating high flows and supplementing low flows (Mbona, 2016). Referring to a study on Afromontane wetlands, Chantanga *et al*. (2020) related the wetland altitude to the ecological services from a wetland. High-altitude wetlands were linked to streamflow regulation. In terms of the whole Krom River, the case study wetland is located in the headwaters making it highly likely for this wetland to regulate streamflow. In terms of research in the Krom catchment, other studies have reported strong flood control from the palmiet wetland (Ellery and Kotze, 2009; Rebelo et a., 2015, 2019; Grundling *et al*., 2017). Concerning the size of the case study wetland, although it accounts for 3 % of the catchment area and is relatively small, a large contributing area (including the surrounding TMG aquifers and alluvium; and rocky terrain with shallow slopes supporting more runoff inputs to the wetland) and the riparian nature of the wetland further supports the likelihood of the wetland's streamflow regulation abilities (Blanchette *et al*., 2022).

Simulating streamflow attenuation was possibly captured in all models because of fewer restrictions and processes regulating the wetland inflows. The constituents of wetland inflows are fewer than the outflow composition. Inflows in both tools was composed of rainfall and runoff. Outflows, however, had more components including AET, return flows to the river in both tools, and an additional outflow representing groundwater in ACRU. In both modelling tools, only the runoff inflows to the wetland were regulated while each outflow was regulated. In the CW wetland, regardless of the wetland storage was full, all inflow were entered the wetland. In the RZ wetland, all upstream groundwater inflows entered the wetland storage (specifically, the subsoil) while all surface water inflows only entered the wetland storage if the soil profile was not saturated. In the WL wetland, runoff inflows from the upstream subcatchment only entered the wetland soil profile if it was saturated. In other words, WRSM-Pitman allowed wetland inflows regardless of the storage deficit while the ACRUs wetlands received surface water inflows in accordance with the storage deficit.

In addition to fewer contributing processes for wetland inflows, reviewing the model routing shows that the inflows are indicative of the wetlands attenuation capacity. Intercepted flood waters and storage retention maps directly to attenuation. In other words, inflows to the wetland reflects the wetland's access to water which may or may not be attenuated. Alternatively, outflows are more influential on supplementation abilities of the wetland model since the wetland outflows are related to the downstream measures of the catchment water yield. This is evident in the model setup with the outflows contributing to the catchment streamflow at the outlet in the WRSM-Pitman model and with the wetland outflow representing the catchment outlet in the ACRU models.

Relative to other studies, the consistent attenuation of streamflow could be accepted with caution. The extent and occurrence of streamflow attenuation by a wetland has been related to the wetland storage preceding the flood event (Morris and Camino, 2011; Cui *et al*., 2020). In this case, the greater the wetland storage deficit, the greater the likelihood and volume of streamflow attenuation. Moreover, Morris and Camino (2011) provided motivations for different wetland storage volumes leading to a physical wetland not performing the expected streamflow attenuation recorded in literature or performing a contradictory role. With limited monitoring and data on the wetland storage preceding flood events, it is difficult to conclude whether all flood events in Kromme catchment were attenuated, as shown in Table 13. Even though all floods were attenuated, the possibility for the case study wetland to not attenuate flows in reality and in the models should be tolerated. What is promising, however, from these results is the variable extent of streamflow attenuation for the flood events in each model. This reflects that the wetland storage was changing over time and influencing the modelled streamflow attenuation. In addition to this, there is evidence from other studies for better model performance from conceptual models during wet conditions (Bai *et al*., 2021).

Regarding low flows, simulating streamflow supplementation associated with the unchannelled valley-bottom wetland type depended on whether the models allowed wetland return flows to the river regardless of the wetland storage being full. Similar to flood attenuation, the wetland storage was critical to simulation of streamflow. Framing the supplementation within the fill-and-spill ideology, the wetland models have different spilling regulations. Some wetland models had to be full before allowing outflows to the river. The CW wetland used this subsequent spillage model where the model has to full before it spills. Other wetland models did not need to be full to release water to the downstream river. This was the case for ACRU's wetland models with compartmentalised wetland storage and differentiated runoff outflows. In terms of the wetland storage characteristics, the division of the wetland storage into the soil profile and groundwater storage allowed surface runoff, interflow and groundwater outflows from the wetland (Figure 38). According to the model configuration, surface runoff was modelled when the wetland soil profile was saturated, interflow was simulated delayed stormflow and groundwater was released regardless of the soil profile storage. This allowed the ACRU wetlands to simulate subsurface outflows from the wetland during low flow periods which are often associated with drier soil profiles (viz.

unsaturated wetland storage) which would lead to more soil water retention and less runoff generated (Figure 38). The subsurface outflows from the groundwater storage allowed the wetland models to continue having runoff despite the wetland storage (in particular, the soil profile) possibly being unsaturated. Conventionally, only a saturated soil profile or wetland storage would lead to the generation of runoff.

However, differentiated wetland storage and runoff do not fully explain why the RZ wetland modelled supplementation while the WL wetland did not. From this perspective, wetland AET, another significant wetland outflow, and inflow pathways to the wetland may explain how the models predicted different effects on catchment streamflow. In the ACRU wetlands, water in the wetland storage is not equally available for AET (Schulze 1995 and updates). Inherent in the model structure, percolated water in the groundwater storage is not available for AET. In terms of user defined configurations, the RZ and WL models were setup to have water available for AET decreased down the soil profile. Since the WL model received all surface and groundwater inflows from the upstream subcatchment through the wetland surface, more water was available for AET as larger volume resided and drained through the soil profile. Alternatively, the RZ wetland received a large proportion of the total inflows (i.e. upstream groundwater) in the subsoil where the water availability for AET is relatively low compared to the topsoil. A saturated subsoil in the RZ wetland led to greater percolation of water into the groundwater storage zone. This resulted in a lower water availability and subsequent AET from the RZ wetland (Figure 33). In addition to this, the HRU's are designed to release larger groundwater outflows with greater volumes of water in the groundwater storage. This is evident with the RZ simulating greater streamflow volumes for all drought periods (Figure 38) and may also explain why the RZ model over simulated the catchment streamflow (Table 11). In combination of these factors (i.e. water availability for wetland AET, different inflow pathways and compartmentalised wetland storage), the final result was the prediction of streamflow supplementation in the RZ wetland with lower water use and a greater groundwater storage volume.

Identifying streamflow supplementation dependent of the wetland storage volume suggests that a fill-or-spill model could be added to the fill-and-spill paradigm. At the fundamental level, implementing a wetland model in hydrological modelling tools involves a wetland water balance represented as a storage unit, tank or bucket which receives and loses water (Savenije, 2010; Rahman *et al*., 2016). Some studies have proposed extensions and variations to the fill-and-spill paradigm to be more inclusive and representative of the processes associated with the particular wetland type and environmental conditions. For geographically isolated wetlands, the fill-and-spill model was modified to fill-spill-merge-and-split (Nasab *et al*., 2017; Lane *et al*., 2019; Evenson *et al*., 2016). For riparian wetlands, the fill-and-spill model applies (Nasab *et al*., 2017; Lee *et al*., 2018b). In the South African context, fill-and-spill is applicable for perched wetlands, riparian or geographically isolated which have a restrictive, underlying substrate (Melley *et al*., 2017). For South African wetlands with seepage due to no restrictive layer, Maherry *et al*. (2017) found that hydrological models of wetlands need a fill-seep-and-spill concept to improve process representation and model performance. In the case of the palmiet case study wetland modelled in this research, there was a restrictive layer underlying the wetland qualifying the spill-and-fill concept as acceptable (Figure 7). Capturing this affected the wetland models ability to simulate streamflow supplementation with examples from ACRU's RZ wetland demonstrating filling-and-spilling and streamflow supplementation. Alternatively, the CW wetland demonstrated filling-or-spilling (i.e. only one process at a time and spillage restricted to saturated storage conditions) and no subsequent streamflow supplementation.

On one hand, the spilling process of streamflow supplementation could be viewed as a threshold process which occurs within some limits of the wetland storage, and that only occurs when other water demands are satisfied. Examples of water demands from the wetland include AET losses from the vegetation, water movement according to the hydraulic gradient between the wetland storage and alluvial fan along with wetland outflows generated when the wetland is relatively saturated or when the rainfall intensity exceeds the soil infiltration rate. Under low-rainfall and drought conditions, the lowering water table, implying less water stored in the wetland, could reduce the extent and likelihood of streamflow supplementation. This was the case for the WL wetland model with high water use and low wetland storage volumes resulting in no supplementation being observed. In other words, the water use and storage deficit was greater than amount of runoff released from the groundwater storage or from the soil profile as interflow.

Alternatively, the spilling process for the case study wetland might not be a threshold process and streamflow supplementation, or at least outflows, occur consistently. According to Valois *et al*. (2020), it is not unusual for a semi-arid peatland to have high groundwater inflows and a high annual water table. In this case, the threshold process may not be as influential. There is further evidence for streamflow supplementation and consistent outflows, and ultimately spilling, which is not entirely a threshold process, from the case study wetland in the catchment and wetland characteristics. In terms of the local hydrology, the catchment is a strategic water source area (Nel *et al*., 2013). Regarding water supply, according to Bate and Adams (2000) the Kromme river catchment has been two dams which often receive transfers from surrounding catchments and mountain aquifers. At the same time, the Kromme catchment needs to supply 40 % of the downstream metropolitan's water supply and water transfers to neighbouring catchments (Rebelo *et al*., 2015; DWS, 2016b). This implies considerable outflows from the catchment with low proportions water yield being held in the upstream sections of the catchment for long periods. In terms of wetland inflows, there is an alluvial fan providing subsurface interflows to the wetland along with multiple tributary inflows from the surrounding slopes providing large groundwater inflows to the wetland. In addition to these inflow volumes, there are preferential flows through the wetland supporting the movement of water through the wetland (Tanner *et al*., 2019; Smith, 2019). Making a case for streamflow supplementation as a likelihood in the case study wetland, in the height the 2016/15 drought, no surface inflows were observed from the main channel, but outflows persisted at the wetland outlet (Tanner *et al*., 2019). This likelihood, together with multiple and consistent inflows, preferential pathways in the wetland and a large but limited wetland storage suggests that the case study wetland does not need to be saturated to spill. This concept was represented in the ACRU wetland models with the compartmentalised wetland storage and consistent groundwater outflows. In this case, this suggests that for the case study wetland, and similar unchannelled valley-bottom wetlands with no seepage, the fill-and-spill paradigm is literal and suitable. In the CW wetland of WRSM-Pitman, the wetland model takes on a fill-or-spill (i.e. only releasing catchment outflows when the wetland storage was full) approach which was unsuitable for the physical wetland and modelling streamflow supplementation. In addition to this, the CW wetland model allows wetland AET to persist as long as the wetland storage is not empty. This promotes wetland AET and a larger storage deficit which needs to be refilled before spilling is modelled. Although it is likely that streamflow supplementation was consistent, this does not prove that every drought or flood is influenced by the wetland or that the streamflow regulation is linear (viz. occurring to the same extent for every extreme event).

Considering the timing of streamflow attenuation and supplementation, modelled and expected behaviour from literature are similar. Compared to classification literature, the unchannelled valley-bottom wetlands are equally likely to attenuate or supplement flows in the early and late phases of the wet season. Rainfall is reported to be bimodal in the Kromme catchment (Nsor and Gambiza, 2013). The climate data used for modelling the catchment showed bimodal rainfall distributed in late Winter (August) and late Summer (December) which is slightly later than reported Spring and Autumn rainfall peaks. With flood events observed in September, July, August, and November, three of the events appear to precede the wet seasons and attenuate flows variably (Table 15; Table 17). Within each model, the August streamflow attenuation (viz. the absolute hydrological impact) is moderate (i.e. neither the maximum, minimum nor only minimum in the case of the WL wetland). In terms of droughts, a similar comparison is not reliable or exact because of the padding and durations of SPEI-12 indicating drought conditions longer than a month (Table 18). However, in terms of streamflow supplementation and continuous outflows, monitoring a valley-bottom, peat wetland in the Canadian Rocky Mountains, which is similar to the case study wetland typology, observed that outflows continued during droughts (Streich, 2019). Local monitoring has reported the same occurrence of outflows from the case study wetland during droughts (Tanner *et al*., 2019). This further supports the likelihood of streamflow supplementation during droughts and its predictions from the RZ wetland.

With reference to the long-term model behaviour for the wetland models (i.e. the common, net or likely response of the wetland model over the whole simulation period), the hydrological environment which the models were developed for may have resulted in the similar hydrological impacts on streamflow during droughts. In this case, hydrological impact refers to the streamflow volume difference from the catchment with and without a wetland. Model development efforts as early as the 1980's noted that the distribution of water to different components in the hydrological water balances differs by regions and climates (see UNESCO, 1981). Recent comparative modelling in de Boer-Euser *et al*. (2017) linked model performance to different climates, rainfall inputs, runoff pathways and volumes. Generally, cool, wetter climates are thought of as having more surface runoff and low water use while warmer, drier climates are predominantly subsurface runoff areas with high water use. Both ACRU and WRSM-Pitman were developed and validated in southern African catchments. As a result, the hydrological responses and catchment concepts incorporated in their model structures are relevant and representative of the large-scale semi-arid conditions and a few humid conditions. It is possible that the long-term hydrological impact reflects the model development for drier conditions (Table 13). For each wetland model, the same hydrological impact (i.e. attenuation or supplementation) found in the total catchment streamflow for the whole simulation period was predicted during the drought period assessment (Table 18). For example, the CW wetland attenuated flows during the full simulation period and for all three selected drought periods. The same trend was observed for the ACRU wetlands with the RZ wetland supplementing flows and the WL wetland attenuating flows. These results suggest that the long-term model behaviour is similar to the wetland model responses during droughts. This is an interesting and informative relationship to identify since Dai *et al*. (2010) found similar uncertainty and variability when modelling streamflow from catchments with wetlands under dry conditions.

Another factor potentially influencing the wetland model responses detected in the analyses is the metrics. One would expect that streamflow attenuation would pair with an increase in the wetland storage and streamflow supplementation would pair with a decrease in the wetland storage. This was only the case for streamflow attenuation during floods. However, concerning droughts, the expected pairings were not always the case. One reason is from the contextual differences in the metrics used for identifying the wetlands influence which can result in somewhat counterintuitive hydrological impacts and wetland flux pairings. For example, the hydrological impact could state attenuation while the wetland storage was decreasing. One may expect streamflow attenuation to be associated with an increasing wetland storage which is plausible. However, for the metrics used in this study, the hydrological impact relates to the loss of the wetland storage when comparing streamflow modelled from the catchment with and without a wetland. On the other hand, the wetland storage state relates to the inflows and outflows for the catchment setup with a wetland. The second reason for different pairings is the number of variables included in the metrics. The hydrological impact refers to one variable, catchment streamflow only, while the wetland flux summarises several inflows and outflows of the wetland water balance. For these reasons, during drought periods in Table 18, it was possible for one, the CW wetland to attenuate streamflow and model both a decreasing and increasing wetland storage with AET persisting; and two, for the RZ wetland to supplement the catchment streamflow with consistent groundwater outflows in the scenario with a wetland while the wetland storage increased due to upstream inflows and rainfall to the soil profile exceeding the wetland AET and outflows.

Compared to other studies which used paired scenarios in ACRU4 and SPATSIM-Pitman, model performance without a wetland was found to be better than the model performance from the setup with a wetland (Maherry *et al*., 2017; Mandlazi; 2017). Unlike this research, these two studies parameterised the paired scenarios differently. Moreover, these studies related poor model performance to the hydrological reasoning in the models being insufficient to describe their physical wetlands. In the case of these study's scenarios without a wetland, the models were giving the correct answers for the wrong reasons as over parameterised models to account for the impact of the wetlands. In this research and results on streamflow regulation, the model was first configured with a wetland. The same parameters were used in the scenarios with and without a wetland. Consequently, it is possible that the paired scenario findings in this study are more closely linked to the hydrological processes in the catchment. Model performance in the catchment setup without a wetland was not tested since it does not reflect the actual catchment conditions and the selected land use to replace the wetland area is subjective, uncertain and affects the final hydrological impact of the wetland. In other words, it is not certain whether the replacement land use in the scenario without a wetland should reflect natural vegetation conditions or the competing land uses, which in the Kromme would be agriculture or invasive trees. Furthermore, the hydrological impact will differ depending on the water use and surface properties of the replacement land use: if the replacement land use has a lower water use or rougher surface than the wetland, the absence of a wetland will result in more runoff from the catchment compared to the scenario with a wetland. As a result, the hydrological impact will be lower and vice versa for a replacement land use with higher water use or surface properties than the wetland.

Shifting from particular roles and the selected metrics to the bigger picture, the simulated streamflow regulation roles modelled can be related to the model performance. The results indicate that a model can have moderate model performance and fulfil the regulatory roles of the physical wetland and vice versa. Considering the RZ model, both of the expected streamflow regulation roles were modelled with acceptable streamflow model performance and poor AET simulations. Concerning the WL model, only streamflow attenuation for floods was modelled while there was good model performance for streamflow and AET. If model performance is prioritised together with the fact that streamflow supplementation was possible and likely, model performance does not map well to a model predicting streamflow regulation roles associated with the wetland type. In this case, the WL wetland is the most plausible. However, if model performance is ignored and only the streamflow roles from literature are used as qualifying criteria for a credible model, then the RZ wetland is the most plausible model, and model performance (which was moderate for the RZ model) still does not correlate with prediction of streamflow regulation. Therefore, good model performance does not guarantee the simulation of streamflow regulation roles recorded in literature and moderate model performance does not limit the prediction of expected streamflow regulation roles.

Relating the findings from the modelling to the model suitability detected in the quantitative assessment of wetland representation, both assessments favour hydrological realism. In the qualitative analysis based on wetland characteristics and processes, the wetland model's suitability favoured hydrological realism which was more prominent in the physically-based models. From the modelling in this chapter, successfully replicating the observed data set and simulating streamflow regulation was also observed in the physically-based models: ACRU's WL and RZ wetland's. In terms of the wetland storage properties, both ACRU models conceptualise the wetland storage as a vegetated land mass which is accurate compared to the case study wetland. In terms of the inflow pathways, the RZ wetland achieves hydrological realism by allowing upstream inflows to reach the wetland as subsurface flow and overbank spilling into the soil profile. Hydrological realism is critical for the wetland storage and regulation. Monitoring of several valley-bottom wetlands in Faul *et al*. (2016) related wetland storage to the hydraulic, hydrological and physical properties that promote water retention. In addition to this, the RZ wetland predicted both streamflow attenuation during foods and supplementation during droughts which is expected from unchannelled valley-bottom wetlands. In terms of model performance, the WL wetland simulated the qualifying variables from the catchment and wetland water balance acceptably. According to Evenson *et al*. (2018) this is essential to finding credible models and eliminating models which are acceptable for the catchment but unsuitable for the wetland model. Examples of the over-inclusion were present in this study, similar to the findings of Evenson *et al*. (2018), when considering the acceptable performance of the catchment streamflow in the RZ and CW models and poor simulations of wetland AET.

In terms of the most conceptual model, WRSM-Pitman, hydrological realism was not optimised in the wetland model, affecting the model performance and prediction skill during extreme events. The wetland storage was conceptualised as a single water body similar to a reservoir or lake which is unrepresentative of the physical wetland. Additionally, the parameters were difficult to relate to the available data. For example, how to proportion upstream inflows into water flowing through the wetland and water retained in the wetland storage or the proportion of outflows above the maximum wetland storage that is gradually released to the downstream river. Despite estimating these values by trial and error (viz. calibration) and hydrological reasoning, the model was able to produce an excellent catchment hydrograph for high, low and average flows. Under extreme events, all flood waters were intercepted even if the wetland storage was full which at best simulates delayed streamflow and during droughts supplementation could not be predicted due to ongoing, high AET and storage-restricted outflows. In reality, it is unlikely for the case study wetland to intercept and store all flood waters, especially when the wetland storage is full. Additionally, streamflow supplementation and consistent outflows are associated with the case study wetland. This is not to say that conceptual models are incapable of modelling wetlands, only that the selected conceptual model configured for the case study catchment and wetland in this study did not maximise hydrological realism and this had a bearing on the model performance and simulated streamflow regulation. There are several examples of conceptual modelling tools representing a particular wetland effectively (Savenije, 2010; Hughes *et al*., 2013; Fossey and Rousseau, 2015; Chomba *et al*., 2021; Makungu and Hughes, 2021). In terms of the challenge associated with parameters having minimal physical meaning, what stands out from these studies is the need to develop parameter guidance by applying the wetland model to several different types of wetlands. This approach could be exceptionally beneficial to the use of the CW wetland model.

### 4.4.2. Limitations

The main limitation to the previous results presented and discussed is the scope of the assessment. The analyses and modelling considered one wetland type applied to three models from two modelling tools. In reality, there are six other HGM wetland types, several modelling tools and infinite model configurations that can be explored. Within the scope of this study, the modelling tools MIKE SHE coupled with MIKE Hydro and SWAT are yet to be configured for the case study wetland and have their modelling potential assessed. In terms of the configurations for the selected modelling tools, alternative parameters and models can be setup. For example, in the RZ model, the distribution of upstream baseflows to the wetland subsoil could be adjusted with the aim of reducing the over simulated catchment streamflow and improving the model performance. In terms of model performance from the ACRU wetlands, the legacy errors from the daily simulations propagated into the monthly simulations. These setups have room for improvement. Regarding the CW wetland configuration, the wetland was set to release all water above the maximum wetland storage in the next month. In addition to the fill-or-spill regulations in the model, this instant release of excess storage may have limited the model's ability to simulate supplementation.

In terms of the model accuracy, equifinality is an issue which also needs to be considered from the results obtained in this study. It is possible for several models to yield equally good or similar results. This means that the models used in this study are not absolute. Alternative models and parameter combinations for the Kromme catchment, especially the wetland, could be developed and be equally or more plausible than the models used in these analyses.

In terms of the model configuration, a few key processes, interactions and catchment properties were not taken into consideration. House *et al*. (2016) proposed that wetland management practices and surface water-groundwater interactions are part and parcel of the hydrological processes in a wetland. In terms of processes considered in the modelling, groundwater flows and surface water-groundwater interactions were not explicitly represented nor fully considered. In terms of catchment properties changing over time, the distribution and density of palmiet in response to encroachment or clearing of invasive plants, fires, major floods and the subsequent channel erosion, changes in land use or management factors in the surrounding catchment were not considered in the model input or simulations. Moreover, cycles of cutting and filling from gully erosion led to palmiet formation with channel widening which, in turn, promotes palmiet colonisation (Pulley *et al*., 2018). Changes in the wetland slope, depth or area as a result of this process were not factored into the simulations. Additionally, any changes to the wetland area, depth, slope or vegetation and upstream sediment accumulation in response to the construction of a gabion weir further downstream were also not considered in the model setup and output. Currently, these exclusions are valid concerns since there is an example from dynamic and scenario modelling on wetland filling which reported increased inundation extents of wetlands in response to sediment retention (Copp *et al*., 2007). Finally, all wetland depths used in the modelling study were from field campaigns conducted in 2017 (viz. before the 2018 flood, after the previous floods and gabion weir installations). Wetland depths pre-dating 2017 were assumed to be applicable and representative of properties outside of the monitoring even though there could have been some changes in response to the natural and anthropogenic interferences.

The next noteworthy limitation to this modelling study was the observational data (see section 4.4.1.1). Regarding streamflow data, a short data set was available and derived from piezometers which are suboptimal gauging structures compared to weirs and stage. Streamflow measures from transducers are less accurate than weir and flume recordings (Dobriyal *et al*., 2017). Relying on this streamflow to verify model performance, the modelling study assumed that model performance obtained in the calibration period was conserved, or lower, in the assessment period. There was no streamflow data available for the assessment period which means model performance was not accounted for. Most modelling studies show model performance decreases when applied to data outside of the calibration period. This may reduce the finality and confidence in the results from the assessment period. Concerning the AET data, remotely-sensed products have been reported to under simulate AET due to pixels including land uses surrounding the wetland and water use dynamics from the palmiet physiology in response to the climate which not captured in the current calculations (Ramatsabana *et al*., 2019; Rebelo *et al*., 2019b). Fortunately, the extent of AET under estimation is lowest from the FruitLook product (Ramatsabana *et al*., 2019). In addition to this, remotely-sensed data was summed up from 8-day estimates to monthly values. Therefore, some data points within in month may have occurred in the previous or following month while modelled data was exclusively for a specific month. These observational datasets, with some limitations, were used in the model performance assessments to validate the wetland models. Regarding general data quality, systematic errors in the any observational dataset have been widely flagged and, as a result, qualifying models as acceptable remains challenging and uncertain as we rely on imperfect data to define model performance (Beven, 2018).

Furthermore, data describing and monitoring the wetland storage was limited. In terms of wetland storage, the volume was required as input to the models and monitored wetland storage could be used to establish the model performance. The input wetland storage was estimated from the porosity and soil depth, and it is likely to not be equivalent to the wetland storage in reality. Whether the estimated wetland storage was less or more than the actual wetland storage volume is unknown. For establishing model performance, wetland storage is not monitored at this site. Wetland storage, similar to catchment streamflow, reflects the integrates the inflows and outflows to the wetland giving a broader perspective on the wetland model than AET which is one component of the water balance. Fortunately, AET is usually a large component of semi-arid water balances. However, the microclimate of the case study catchment is temperate. Furthermore, acceptable wetland AET does not guarantee that the other components of the wetland water balance were simulated accurately. Moreover, as a vegetated land mass, efforts to remotely monitor wetland storage in the case study wetland, and similar wetlands which are not water bodies, is still severely limited (DeVries *et al*., 2017).

Concerning the analyses and methodology, assumptions about the model performance and catchment properties together with the metrics selected may be another limiting factor of the study. As previously mentioned, the model performance in the calibration period was assumed to apply to the assessment period. In addition to this, the catchment properties were assumed to remain consistent over the modelling period. However, from 1950 to present date the land use has changed significantly in terms of wetland degradation and rehabilitation, cycles of invasive trees infestation and clearing, and changes in the agricultural cultivation and extent (Rebelo *et al*., 2015; van Wilgen *et al*., 2012; Working for Wetlands; 2005 and updates). This is in addition to several major floods which have impacted the river floodplain and wetland extent and catchment runoff responses to rainfall in the K90A catchment (larger extent of the case study catchment) (Kotze and Ellery, 2009; Rebelo *et al*., 2015). These changes and impacts in the upper K90A catchment were not considered in the modelling. Considering the selected metrics, the hydrological impact and change in wetland storage were interpreted as absolute (i.e. either attenuating or supplementing streamflow and increasing or decreasing wetland storage) at the monthly time step. Although the results had to be upscaled to the largest time step for the selected models and to be less dependent on daily flows with low confidence, the assessment of floods could be performed on an event basis (e.g. over the days which the rainfall event occurred).

In terms of model specific limitations, the tools reconstructing the water balance in the tools is challenging and the simulations rely on static calculation procedures or parameters. In WRSM-Pitman, wetland inflows from the river, return flows to the river and storage can be exported from the model. However, monthly variable properties were not provided and had to be estimated outside of the model (e.g. the average wetland surface area for the wetland storage, rainfall and AET). The long-term water balance for the entire simulation period was equivalent to the change in wetland storage from the model output. However, on a month-to-month basis, there were occasions where the calculated value outside of the model was different from the model output. As such, reconstructing the water balance is time-consuming and values may be different, in some cases, from the models internal calculations. In terms of the model structure, the CW wetland in WRSM Pitman was limited by its approach for estimating AET. In this model, wetland AET was taken as the long-term monthly pan evaporation and coefficient inputs which were repeated for every year in the simulation period. Although this represents the long-term averages, the approach neglects the interannual climate variability and the changes in AET for dry and wet years. This is particularly limiting during dry conditions where AET simulations persist at the local average rate. In addition to this, the AET calculation method in the CW wetlands uses substandard reference evaporation. The wetland routine relies on A-pan and S-pan factors even though the FAO-56 Penman Monteith method (Allen *et al*., 1998) has become widely accepted as the best standard. Understandably, the A-pan and S-pan dependence supports the existing data sets and monitoring efforts in South Africa.

In ACRU, many outputs can be exported. However, the documentation supporting the output options, conversions and water balances is minimal and is yet to be updated. In terms of the model structure, the groundwater outflow was defined as a proportion of the current volume. This value was calibrated and remained consistent for the simulation period. Moreover, it has no physical meaning that can be related to the hydraulic conductivity of the wetland storage. Groundwater dynamics in the model are important and influential for the representation of the physical wetland which is largely groundwater fed and mostly subsurface outflows. Furthermore, in line with limitations of the modelling methods, the impact of changes in land use and human influences, and their impact on land, river or wetland properties affecting streamflow regulation, were not considered when setting up the models. Other limitations beyond the scope of this research was the interaction of hydrological drivers in affecting runoff and the degree of non-linearity in runoff generation processes and streamflow regulation by wetlands.

In both modelling tools, ACRU and WRSM-Pitman, water availability for AET was slightly questionable. In ACRU, the deeper the water in the wetland storage, the less available it was for AET. In WRSM-Pitman, surface and groundwater inflows all received as surface water inflows and stored in a single wetland storage unit was always available for AET. In addition to this, the wetland AET was provided as a bulk-term such that it was not differentiated into soil water evaporation and transpiration which occurs from an unchannelled valley-bottom wetland. According to the comprehensive configuration of hydrological processes in a wetland water balance in Wolski *et al*. (2006), groundwater is often available for AET. The same understanding is embedded in WRSM-Pitman's model structure for the catchment but excluded from the wetland concept. Moreover, the deep roots of palmiet (rooting depths upto 5 m and stem diameter of 0.15 m) in the average wetland profile depth of 6.8 m (Tanner *et al*., 2019; Lagasse, 2017; Pulley *et al*., 2018) suggest that subsurface water would be available to wetland vegetation. In the case of WRSM-Pitman, the representation of wetland AET without limits from soil water content, plant stress or in response to temporally variable climate conditions is unrealistic. As a result, the representations of groundwater AET were inadequate in wetland models.

Lastly, in terms of streamflow regulation, the thresholds governing the wetland responses are currently unknown. On one hand, it is not clear when streamflow regulation is performed by the wetland and when to expect it from the models. The results from this study were made under two main assumptions. The first assumption was that if the catchment streamflow and AET model performance were acceptable then the model was plausible. The second assumption the study operated under was that the classification system highlighted the hydrological roles expected from the wetland type and the models. However, it is possible for a wetland to have no impact on the catchment streamflow (Riddell *et al*., 2013), or to have a role contracting the expectations in literature or to occasionally, as opposed to always, perform streamflow regulation (Camino and Morris, 2011; Kadykalo and Findlay, 2016), or to have a threshold related to wetland storage and climate where streamflow regulation is initiated or terminated (Salimi et al., 2021). On the other hand, when streamflow regulation does occur from the wetland, it is not yet known, nor was it considered in this study which wetland properties would regulate the extent of the streamflow attenuation or supplementation. Examples of potential wetland properties include the thresholds imposed from the soil and wetland storage, how water inflows are distributed to the alluvial aquifer surrounding the wetland or at what wetland storage levels wetland AET and streamflow supplementation stop occurring simultaneously under drought conditions.

### 4.4.3. Implications

#### 4.4.3.1. Modelling and water management

The findings from this study suggest that the model selection process for catchments with wetlands should prioritise hydrological realism in the wetland model. Better model performance and the simulation of expected streamflow regulation was found in models with more in common with the physical wetland. In the qualitative analysis of wetland representation, the compatibility score increased for models which incorporated the physical wetland's fill-spill design and storage properties. In the quantitative analysis, the wetland models with compatible storage properties, water-use and inflow-outflow regulations had better model performance and regulation roles predicted. As a result, it is evident that insufficient process coverage leads to poor model performance and inadequate wetland conceptualisations. Maximising hydrological realism is a step towards model predictions being simulated for the correct reasons.

In terms of the relationship between model performance and streamflow regulation, good model performance does not guarantee the simulation of streamflow regulation associated with the wetland type. Generally, streamflow simulations are the standard measure of model performance. Acceptable streamflow simulations did not lead to good predictions of AET. Moreover, acceptable streamflow and AET simulations did not lead to the simulation of the expected streamflow regulation from an unchannelled valley-bottom wetland. Rather, the model with moderate model performance simulated the expected streamflow regulation roles for the case study wetland. This suggests that it is critical to know what streamflow regulation roles are expected from the case study wetland. More research and monitoring are necessary to define whether model performance or streamflow regulation can equally qualify a model. More importantly, the differences in model responses and absence of model performance leading to the simulation of streamflow regulation highlights that it is essential to know the characteristic responses from the wetland model for the whole simulation period and under extreme events.

When looking to define the characteristic responses from the wetland, modellers can define the streamflow regulation of the wetland model by investigating the model's response to droughts and floods. In terms of metrics, the change in wetland storage has several advantages over the hydrological impact. The change in wetland storage considers the model setup with a wetland which is more realistic and independent of the replacement vegetation in a scenario without a wetland. In addition to this, the wetland storage wetland incorporates hydrodynamics by illustrating inflows and outflows to the wetland.

Furthermore, caution is necessary when assuming a water surplus or deficit from the model output. Wetlands are ecological infrastructure supporting local water security. This is accepted in public, citizen science (Bonthuys, 2020) and academic spheres (Mander *et al*., 2017; Nel *et al*., 2013) with modelling used to guide the estimation and management of water supplies from wetlands. This modelling study demonstrated the possibility for models to predict different water yields, streamflow regulation roles and extents which could result in different estimates of the long-term water supply. Different simulated streamflow volumes may misinform water resource decisions. In one case, under simulated water yields may initiate drastic measures to conserve water resources in the downstream area, limiting socioeconomic development and increasing intermittent water supply (Loubser *et al*., 2020). This could also lead to underestimated ecological reserve flows. Contrastingly, over simulated water yields may exacerbate water insecurity by over allocating water resources which are not available in reality. Inaccurate water yields are high risk since 98 % of South Africa's water resources are already allocated (Hedden and CIlliers, 2014). In terms of competing land cover and uses, the case study wetland is currently threatened by encroaching alien vegetation and agriculture. Black wattle has been reported to use large volumes of groundwater (Ngubo *et al.*, 2022; Scott-Shaw et al., 2017). Over simulated catchment streamflow would underestimate the impact of these activities (i.e. invasive vegetation and agriculture) and overestimate the water available for downstream users. In terms of streamflow regulation, some of the results did not predict the expected roles. Caution is also necessary when assuming that there is no streamflow regulation from modelled wetland or that the regulation is consistent (viz. does not change over time, with degradation or happens with every event for the full duration of the event). In terms of confidence in the results, there is currently no information to substantiate what roles were performed in the selected events. This further highlights the need to be cautious when accepting modelled streamflow regulation and the extents.

#### 4.4.3.2. Landscape management

Another area where different simulated responses could have a significant impact is in scenario modelling for water resource and landscape management. Modelling studies using scenarios could be biased by the model's tendency to give specific responses. In this case, it is critical to establish the simulated wetland's behaviour before interpreting or applying the results when using hydrological models in restoration and impact assessments.

Additional caution is necessary when using the model results to imply or inform ecosystem services related to streamflow regulation. Concerning the landscape, hydrological models are often applied to impact assessment regarding climate and land use change. The results from these studies may be used to advocate for different actions in the catchment. Similar to water management, results from modelled wetlands should be used cautiously or comparatively before informing land planning or restoration. Concerning wetlands, the impact of different model predictions should also be considered when economically valuating ecological system services or implementing associated payment systems if models are used to determine either (Brander, 2013; Rebelo *et al.*, 2017). Variable results from models and unknown extents of streamflow regulation in reality may make it difficult and imprecise to implement payment of services or fines. At this stage, it may be safer to work with the type of role performed instead of the extent of a specific role.

Regarding the downstream water quality which wetlands may improve, the selected models can imply different treatment requirements and sediment loads. One of the ecosystem services provided by the case study wetland is water purification. The palmiet wetland has been reported to provide moderately high water purification roles (e.g. toxicant, Nitrate and Phosphate removal) (Grundling *et al*., 2017; Curmi *et al*., 2019; Rebelo *et al*., 2019a; Lee et al., 2020). This aligns with the findings for low-altitude wetlands to contribute to water purification and erosion-control (Chantanga *et al*., 2020) whereby the case study wetland is located in the valley. Previous studies on the K90 catchment associated the wetland state (degraded or pristine) with the level of water purification performed and implications on water treatment: intact, healthy wetlands result in higher water quality and less water treatment requirements downstream (Rebelo, 2012; Rebelo *et al*., 2015). In terms of the results in this study, the different water yields from each model may inform the water. The concentration of nutrients and pollutants varies with the volume of water present. Lower water volumes increase the concentration of particulate matter in water. According to Rebelo *et al*. (2019a), the efficiency of the wetlands water purification leads to different water treatment requirements at the downstream dams. By extension, models which over simulated streamflow are likely to underestimate the contaminant concentrations in the river and subsequent water treatment measures. The opposite could be expected for the WL model which under simulate streamflow. This could have significant impacts since poor water quality is a health concern for communities and cities downstream. From an infrastructure maintenance perspective, high nutrient concentrations can promote algal blooms in dams which are difficult to control.

Another ecosystem service related to water quality and affecting the downstream water yield is sediment trapping provided by the palmiet wetland. Different water yields from the selected models may imply different sediment loadings in the downstream flows. Models which overestimate the catchment streamflow may imply higher sediment loads and vice versa. In addition to this, palmiet wetlands, with their peat beds, are prone to fires. Recent dry conditions have exacerbated their rate of burning (Grundling *et al*., 2021). The ability for these wetlands to attenuate high flows and trap sediments would be lost with the loss or degradation of the wetlands or with hydrophobic soils post-fire (Nasirzadehdizaji and Akyuz, 2022). This would ultimately increase the sediment load transferred downstream. High sediment yields in dams reduces the dam capacity and catchment's ability to supply the necessary water volumes estimated during planning.

#### 4.4.3.3. Biodiversity distribution and management

The protection of life below water is one of the 2030 Sustainable Development Goals (WHO, 2015). Simulations from hydrological modelling highlight likely hydrological flows and conditions which can be related to the distribution of plant and animal communities, and to inform ecological management or rehabilitation actions (House *et al*., 2016; Haapalehto *et al*., 2011; Murray-Hudson *et al*., 2006; Wolski and Murray-Hudson, 2006). In terms of this modelling study for the Kromme catchment context, the modelled runoff output infers different thermoscapes (i.e. water temperature profiles) potentially affecting fish populations. Changing temperatures with wetland saturation and runoff composition can affect the fish and plant biodiversity. Dick *et al*. (2018) showed that saturated wetlands are more connected to the river network. Large subsurface outflows from their case study riparian peatland resulted in higher stream temperatures, while groundwater outflows contributed cooler waters in summer and warmer waters in winter. Native fish species in the Kromme river (i.e. endemic redfin fish) are currently competing with invasive, non-native rainbow trout (Shelton *et al*., 2016). Fish have species specific temperature preferences and sensitivities. Redfins are generally cool water fish with warm-water, drought tolerant species in the Eastern Cape (Reizenberg *et al*., 2018; Reizenberg, 2017). Rainbow trout are also cool water fish but have a higher tolerance window for extremely low temperature waters and intolerance for high temperatures compared to redfins (Chen *et al*., 2015; Melendez and Mueller, 2015; Verhille *et al*., 2016).

Extrapolating the findings of the aforementioned studies on the water temperature preferences for native and invasive fish species to the modelling results of this research, there is a chance for warmer waters to be associated with models simulating large volumes of subsurface runoff compositions (i.e. WL models) and winter or dry season groundwater outflows (i.e. RZ and WL models). Actual thermoscapes of wetland inflows and outflows are yet to be determined for the case study wetland modelled. Although there are direct measures of estimating the river and wetland thermal conditions, using these models to inform conservations measures, threats and fish populations suggests warmer river waters. By implication, higher water temperatures would potentially eradicate rainbow trout invasions followed by the loss of native fish species. Regarding the current climate change projections for the region, a similar scenario is probable in a warming climate. However, Eastern Cape is part of the transition zone between a wetter eastern and drier western South Africa such that climate variability could progress to warmer or cooler temperatures (Mahlalela *et al*., 2020). In the likelihood of a cooler climate, the rainbow trout populations would expand, and the critically endangered status of redfins could be exacerbated. Similar implications cannot be derived from the CW wetland where wetland outflows are not differentiated into surface and subsurface runoff.

Another Sustainable Development Goal related to biodiversity is the preservation of life on land (WHO, 2015). In this case, different water temperatures, based on runoff compositions, could infer different plant communities since nutrient availability changes with water temperature. Plants generally have different nutrient requirements and uptakes nutrients in specific forms – often referred to as bioavailable nutrients (Adamo *et al*., 2014). Therefore, these changes in the chemical composition of the wetland or river water could promote different plant communities. Varied plant communities are critical to modelling the Kromme catchment accurately where palmiet is central to the wetland's formation and streamflow regulation abilities (Williams, 2018; Grundling *et al*., 2017). In terms of model algorithms, variable plant communities are critical because of the variability in the model's simulations of wetland AET.

### 4.4.4. Recommendations for future research

Following these results and implications, there are opportunities for future research to advance the current study. First and foremost, increasing the scope of the study is necessary. This involves modelling other wetland types in several catchments and of different sizes. Alternative model setups could be investigated, and different modelling tools could be assessed. Diversifying the model setups, tools and wetland types investigated would test the suitability of several models and identify conceptual models and configurations for each wetland type (i.e. build a repository of working models and parameters for specific wetlands and conditions). For continuity, these recommendations could continue to explore whether the qualitative analysis of wetland representation with compatibility scores correlates with model performance on a wetland type basis. It could also highlight which criteria in the qualitative analysis are missing or irrelevant for other wetland types. In addition to this, increasing the scope of the study provides more evidence for whether model performance guarantees the simulation of expected streamflow regulation roles from the wetland. Finally, the current case study focused on a small wetland and catchment. More studies and modelling for catchment scales used in water resource management would be beneficial.

Secondly, investments into monitoring wetlands will be essential to facilitate modelling wetlands with some degree of confidence (viz. an assurance that the processes in the model and subsequent results are correct) and knowing which streamflow regulation roles to expect from the wetlands. One way of addressing uncertainties and limited data would be to prioritise long-term monitoring. The effective modelling of wetlands will be supported by continued monitoring of wetlands, and for a wider range of variables. Monitored variables, ideally, would include wetland inflows, outflows, the thresholds of streamflow regulation and the storage properties and volume. It would also be helpful to address challenges or methods for upscaling the wetland observations from the site to catchment scale. Together, these long-term datasets for several variables would help to confirm which streamflow regulation roles to expect from the models and to validate the wetland models. In the bigger picture of conservation and large-scale remediation measures imposed with permanent effects, such as the efforts and costs invested into the instalment of erosion structures in the upper Kromme, it should be mandatory to include maintenance and monitoring devices to improve the understanding of the wetlands and monitor the effects of the interventions.

Related to monitoring, the third recommendation for future research comes in the form of investments into accurately representing the vegetation distributions and water use. In terms of the vegetation distribution, a previous study found that the Kromme catchment runoff changed with changes in land use (Rebelo, 2012; Rebelo *et al*., 2015). From the current modelling, streamflow volumes and regulation were sensitive to the AET algorithms; therefore, land uses and covers in each model. One way to address this would be to incorporate the land cover changes over time in Kromme catchment into the simulation period. This means separating the simulation period into phases representing the dominant land use or cover before major interventions (i.e. pristine conditions, degraded state and the catchment after clearing alien vegetation to the current re-establishment of alien vegetation). If modelling the larger catchment, one could also divide the simulation period by according to changes form the introduction of gabion weirs or before and after major flood events that altered the floodplain. This recommendation may answer whether capturing these changes improves model performance. In terms of addressing the water use, another way to address its representation could be to relate the ground-based AET to wetland outflows. This would be particularly helpful for understanding the thresholds of streamflow supplementation, especially during droughts. This may also help to determine whether the streamflow regulation roles can verify a model in the same way as model performance.

Outside the scope of modelling, model developments and improving the ease of water balance accounting in modelling tools will improve modelling and validating catchments with wetlands. In WRSM-Pitman, revisions to the estimation of AET may improve the wetland's model performance. It may also help to incorporate more fill and spill options to the CW wetland model structure to represent an unchannelled valley-bottom wetland. For example, separating the wetland storage into a surface and subsurface compartment, allowing wetland outflows to the downstream river despite the wetland storage not being full and reducing the wetland AET as the wetland storage decreases. Working with the current model structures for both modelling tools (i.e. ACRU and WRSM-Pitman), building a knowledge base of working models and parameter sets for specific wetland types would be helpful.

In terms of the models ease of use, improvements can be made to the process for estimating the water balance. Estimating the water balance adds critical context for interpreting the wetlands behaviour and investments could be made to output the associated variables. As more modelling studies move towards estimating the water balance (Ghazal *et al*., 2021; Muhammad *et al*., 2019), it is likely to become a standard for modelling studies. This is in line with the shift from validating streamflow to ensuring integrity for the catchment water balance (i.e. model evaluation) (Euser *et al*., 2013; Gupta *et al*., 2013). The current output options from ACRU and WRSM-Pitman do not output a synthesised water balance. Both modelling tools may consider adding a water balance utility in upcoming versions. In addition to this, the model documentation could be revised and updated. These recommendations will help to easily synthesise model output, validate wetland models as well as represent and configure several wetland types and their dominant wetland processes in the respective tools.

There is also scope to investigate whether wetland representation improves with increasing detail in modelling the groundwater and river dynamics. Hydrological models have been noted for their limitations in over simplified modelling of rivers (Chomba *et al*., 2021) and groundwater (Acreman, 2007). These setbacks have been addressed by coupling hydrological models with hydraulic models for floodplain wetlands (Chomba *et al*., 2021; Makungu and Hughes, 2021) and groundwater models capturing the TMG aquifer dynamics (Watson, 2018). It is possible that capturing these influences in the groundwater-dependent Kromme catchment, river and wetlands may improve the modelled wetland inflows and flow pathways between the river and wetland.

In terms of analysing the modelled results, more metrics and variables could be included or a standard method for impact detection could be introduced. There are many ways of identifying a wetland's impact on the catchment streamflow. Other metric options include extreme flows associated with return periods, percentiles, base flow index, wet versus dry year responses or analyses relative to the growing season where water abstractions may increase and affect streamflow regulation. In terms of the temporal scale of the metrics, the floods could be assessed within the event duration instead of the month in daily models. Although there are many options, Kaykalo and Findlay (2016) reported that consensus on the hydrological function of wetlands is confounded by the different metrics used in different case studies. In this case, it would be beneficial to embrace continuity: choose a suite of metrics for identifying a simulated wetland's impact on catchment streamflow. This would keep the metrics standard and allow previous and upcoming studies to be comparable. The hydrological impact and wetland storage flux have shown potential to be a simple starting point for the proposed suite of metrics. Moving forward, it would be useful to investigate which metrics would be most useful, needed or used by modellers and water resource planners and compare these values from the modelled catchments with wetlands.

### 4.4.5. Concluding summary

Assessing the model output revealed that with stricter model performance standards fewer models remain as viable representations of the wetland. This chapter showed that acceptable streamflow model performance for an unchannelled valley-bottom wetland can be achieved in different modelling tools and models although the streamflow volumes vary between different models. Some models over simulate the catchment streamflow while others under simulated streamflow. In terms of stricter model performance standards using the catchment and wetland water balance, the inaccuracies and variability in simulated wetland evapotranspiration became obvious. It was possible for models with good streamflow performance to have unsatisfactory model performance for wetland AET, making one out of three wetland models credible according to the standard of good model performance for a catchment and wetland water balance variable. In terms of simulating the streamflow regulation expected from an unchannelled valley-bottom wetland, all models demonstrated streamflow attenuation for floods. During droughts, one wetland model simulated streamflow supplementation. This was attributed to the ability of the model to release consistent groundwater outflows, low evapotranspiration and the allowance of wetland outflows despite the wetland storage not being full.

Whether comparing models using model structure concepts or applying quantitative modelling, wetland representation improves with the extent of similarities between the physical and model wetland. Modelling the catchment revealed that maintaining a high level of similarities between the physical and simulated wetland increases the likelihood of good model performance and the simulation of streamflow regulation expected from the case study wetland. This was also evident in the qualitative assessment of wetland representation based on the wetland model's characteristics and processes. In this case, compatible wetland models had properties expressing the physical wetland.

However, acceptable model performance did not guarantee the simulation of all expected roles. This was the case for the wetland model passing the model performance tests for the catchment and wetland water balance variables but only simulating one of the expected streamflow regulation roles. Similarly, it was possible for a model with moderate model performance to simulate both of the streamflow regulation roles expected from an unchannelled valley-bottom wetland. This is a slight conundrum, and one modeller's should be cautious of. At best, and depending on the data availability and quality, there appears to be a need to decide whether the model is credible based on the model performance or streamflow regulation or both. The modelling in this study supports considering whether streamflow supplementation from the wetland during droughts is an occasional possibility.

Moving forward, there are differences in model output which could lead to different decisions. Setting up the same wetland in different modelling tools with the same climate input gave different streamflow volumes and responses to extreme events. The results demonstrated the possible variability in water yields and streamflow regulation roles embedded in a wetland model. These differences need to be considered before applying a model's output to resource use, management and planning decisions or impact assessments.

More importantly, for any model, it is imperative to identify the long-term, drought and flood streamflow responses since setting up the same wetland in different tools with different flow pathways resulted in different responses. Knowing how the model is likely to respond to extremely wet or dry periods contextualises the streamflow output and wetland's influence on the catchment hydrology. This information may be essential to modelling applications using historical or future climates and impact assessments.

Repeating responses means that the model behaviour can be characterised from past events and used to identify the wetland's influence on the catchment hydrology. Although the actual volumes of streamflow attenuated or supplemented differ, each model showed the same type of response to extreme events (e.g. consistently attenuate flood events or droughts or consistently supplement low flows during droughts).

In terms of model skill during extreme events, the modelling tools show potential for hydrological modelling in catchments with wetlands. The current model structures and configurations used in this study were well equipped to capture streamflow attenuation. In terms of low flows, streamflow supplementation during droughts was variable but the hydrological impact from the simulation period was the same as the model's hydrological impact during droughts. This trend was attributed to the modelling tool's development and verification for the local hydroclimate. Following a few minor updates to the wetland storage properties, units and outflow allowances (both AET and streamflow), improvements can be made to the models' suitability and performance for unchannelled valley-bottom wetlands.

# Chapter 5: Conclusions

## 5.1. Synthesis of findings

This chapter concludes the dissertation on comparing wetland models with a summary of the key findings relative to the research questions, objectives and aim. This is followed by a brief discussion on the implications of the findings, limitations of the study and recommendations for future research.

### 5.1.1. Relative to the research questions

The unchannelled valley-bottom wetland was conceptualised differently in the selected modelling tools and models. The first research question was “how do different modelling tools conceptualise an unchannelled valley-bottom wetland?”. The simulated wetlands compared can be separated into two groups: those compared based on wetland model features and a subset for which the model was applied quantitatively. From the simulated wetlands not modelled, suitable and unsuitable wetland options were present. A suitable wetland model was derived in MIKE SHE coupled with MIKE Hydro River. Here, the case study wetland was conceptualised implicitly, based on low elevations and zone specifications, as areas where water accumulation and channel overflows were permissible. In SWAT, two simulated wetlands options were identified but unsuitable for the riparian case study wetland due to incompatible wetland typologies, processes and storage substrate as open water in the models. Applying two of the tools quantitatively, three simulated wetlands were investigated: 1) ACRU’s wetland HRU where all upstream groundwater outflow was routed to the wetland via river flows, 2) ACRU’s riparian zone HRU where all upstream groundwater outflow was routed to the wetland’s subsoil, and 3) WRSM-Pitman’s comprehensive wetland where a consistent portion of all upstream surface water and groundwater was routed to the wetland storage and the remaining portion represented free flowing water through the wetland’s preferential pathways. In ACRU’s wetlands’, inflows were regulated by the soil profile saturation and infiltration while outflows depended on drainage rates and the groundwater storage volume. In WRSM-Pitman’s wetland, wetland inflows were allowed into the wetland regardless of the wetland storage volume while outflows to the river were restricted to the wetland storage being greater than the maximum capacity and wetland AET persisted regardless of the climate and as long as the wetland storage was not empty.

According to the comparison of the wetland units without modelling, the compatibility of a wetland model increases in more physically-based models which have many similarities with the actual wetland. In this assessment, the suitability assessment was based on the wetland’s dependence on the local topography, wetland typology, water balance components, storage properties and streamflow regulation process. The most complex and fine-scale tool, MIKE SHE, yielded wetland models which were highly suitable for representing the characteristics and processes of the physical wetland. This was followed by good suitability for ACRU’s conceptual and moderately physically-based wetlands. In these wetland models, a vegetated land mass and groundwater represented the wetland storage which is similar to the case study wetland. Next, moderate suitability was identified for the comprehensive wetland in the highly conceptual, lumped WRSM-Pitman tool due to the conceptualisation of wetland storage as a single unit of open water where the physical wetland is actually a vegetated soil profile. In addition to this, the comprehensive wetland relied on several conceptual parameters which were not directly related to physical properties of the wetland to describe the wetland's storage regulation. Concerning wetland typology, a key relationship underpinning these findings was that modelling tools have a limited scope of wetland typologies in their model structures. Furthermore, a simulated wetland is a combination of options within the software and modeller choices. Some features of a simulated wetland are not modifiable by the modeller which were referred to as fixed properties. The compatibility of the simulated wetland improved when the fixed properties were compatible with the case study wetland's features.

From the modelling results, the simulated wetlands have different impacts on catchment streamflow in a time and flow specific manner. Predicted streamflow regulation could be moderately explained by the wetland model algorithms and configurations at the monthly time-step while daily simulations were relatively poor. This is attributed to the intricacies of the fill-and-spill process which is yet to be fully captured in the models or fully explainable with the existing data. The second research question of this study was "what impact do the simulated wetlands have on modelled catchment streamflow during the whole simulation period, floods and droughts?". According to the modelled results, the wetland models differ in volume of streamflow simulated along with the type (e.g. attenuation or supplementation) and volume (viz. extent) of streamflow regulation. Comparing the modelled AET to predicted AET of the wetland models, the models only managed to capture the percentile distribution of remotely-sensed, monthly totals of AET. Poor model performance for the monthly distribution of AET was partly because of different water balance and AET algorithms in the wetland models, and the 8-day timestep of AET in the remotely-sensed data. In terms of long-term analyses using the total streamflow for the simulation period, attenuation was simulated by the comprehensive wetland and wetland HRU while supplementation was simulated by the riparian zone HRU. During floods, all simulated wetlands simulated streamflow attenuation. ACRU's wetlands' simulated attenuation based on a soil moisture deficit in the soil and lagged runoff. Alternatively, the comprehensive wetland simulated attenuation because of no restrictions to wetland inflows relative to the existing wetland storage. During droughts, the model responses varied. Predicting streamflow supplementation was largely explained by differences in the wetland models' AET algorithms. Only one wetland model, the riparian zone HRU, simulated supplementation due to continuous subsurface inflows, which were predominantly unavailable for evapotranspiration, and following drainage to the groundwater storage, resulted in subsequently large groundwater outflows. Contrastingly, attenuation was predicted from the wetland HRU because of high evapotranspiration and a large storage deficit. On the other hand, the comprehensive wetland attenuated streamflow during droughts because of ongoing water losses through evapotranspiration that was insensitive to the interannual climate variability which, in reality, would initiate soil water stress on vegetation reducing evapotranspiration. In addition to this, supplementation in the comprehensive wetland was limited by the model's design to restrict return flows to the river when the wetland storage is full and during droughts the wetland had low storage levels.

It was interesting to discover that model performance does not guarantee the prediction of streamflow regulation roles recorded in literature. This study identified the expected streamflow regulation for unchannelled valley-bottom wetland was predicted by the riparian zone HRU which had moderate model performance, while the wetland HRU with good, all-round model performance for the catchment and wetland water balance simulated streamflow attenuation during floods and droughts. These results suggest that one, supplementation could be a threshold process based on competing wetland outflows: evapotranspiration and returns flows to the river; or that the groundwater in the wetland HRU model was under simulated, resulting in less groundwater outflows; and two, different AET predictions affect which net streamflow regulation process occurs during droughts. In the case of the former concerning supplementation as a threshold process and comparing the two wetland models from ACRU, more groundwater outflows from the wetland HRU may have led to the simulation of streamflow supplementation. Considering the different AET processes in the wetland models, streamflow supplementation was largely attributed to the low AET modelled in the riparian zone HRU wetland model. Alternatively, streamflow attenuation during droughts was attributed to the large AET volumes simulated in the comprehensive wetland model of WRSM-Pitman and ACRU's wetland HRU.

Altogether, these findings are similar to several views in the existing body of literature. Firstly, regarding model performance standards for catchments with wetlands, this research highlighted the need to validate the wetland water balance (or variables) by confirming the potential to have decent model performance for catchment streamflow and moderate to low performance for wetland variable. Without considering the wetland water balance or variables, it is possible to incorrectly assume that satisfactory catchment streamflow implies acceptable hydrological processes in the wetland model. This could lead to establishing unsatisfactory wetland models as plausible. Secondly, in terms of literature relating wetland representation to model type and complexity, the results from this study seem to agree with previous research which recommends the use of more complex models (e.g. physically-based and distributed) to improve wetland representation. Compatible wetland typologies, storage properties and regulation processes in a wetland model was associated with the models which had good model performance and predicted the streamflow regulations expected from an unchannelled valley-bottom wetland. Thirdly, concerning model abilities during different flow periods, the variable wetland storage and hydrological impacts on streamflow during droughts from the modelled wetlands agrees with previous studies showing variable responses of wetland-containing catchments during low flows. These results emphasise the importance of flow-specific assessments where a model is subject to irregular conditions and predictions deviate from regular responses. Furthermore, during low flows, a wetland's hydrological role is critical.

Using both comparative approaches (i.e. comparing wetland models without and with quantitative modelling), the comparisons revealed that model suitability and performance strongly depend on hydrological realism. The third and final research question for this research was “how does wetland model suitability, as assessed based on a conceptual review of model structure, compare to quantitative assessments of models' hydrological flux predictions?”. In the comparison without quantitative modelling, good model suitability was found in the wetland models which closely described the characteristics and processes of the physical wetland. From the comparison with modelling, model performance and streamflow regulation expected from unchannelled valley-bottom wetlands was found in the tool’s with better and explicit representations of the case study wetland’s properties.

However, the results from the comparative methods have different contexts. Contextually, the qualitative analysis was limited to the wetland model only and parameter choices or adjustments which could happen in the calibration process were not assessed. More importantly, without modelling, the comparison could not indicate the hydrological role that would be predicted by the wetland model when applied in a particular catchment and climate context. Alternatively, the modelling results incorporated all aspects of wetland representation: the wetland processes, characteristics and the impact on catchment streamflow. In this case, the wetland features acted together and resulted in predictions of the wetland’s influence on water availability. Therefore, in terms of practicality and efficiency, comparing wetland models with modelling is a final and comprehensive indicator of wetland representation. Alternatively, the comparison of wetland models without modelling is a good, initial indicator of wetland representation. Comparing the wetland units without modelling is likely to suggest which models will lead to better model performance before investing in using a modelling tool and setting up a model which are time and resource intensive tasks.

### 5.1.2. Relative to the objectives

In this dissertation, all objectives were successfully addressed. Concerning the overarching, main objective of the research which was to identify the most suitable setup of the unchannelled valley-bottom wetland. In terms of model performance, ACRU’s wetland HRU appears to be a credible model for the case study wetland. In terms of streamflow regulation, the riparian zone HRU which simulated the attenuation during floods and supplementation during droughts expected from the wetland type.

The first objective of the study was to investigate and compare hydrological characteristics and processes defining wetlands. This objective was achieved in Chapter 2 by comparing features of wetland models relative to the palmiet wetland. Wetland representation can be assessed and compared, with a standardised multicriteria method, using the wetland model concepts. According to this assessment, an unchannelled valley-bottom wetland is best represented in detailed models with intended to represent riparian wetlands and having sufficient groundwater components (e.g. runoff, storage and connections).

The second objective of the study was to determine impact of wetlands on streamflow over the whole simulation period and during extreme events (e.g. droughts and floods) *(*viz.). This was achieved in Chapter 3 by modelling the wetlands as functional units in the catchment and detecting the streamflow and storage responses over the whole simulation period, droughts and floods. According to this assessment, different wetland models predict different streamflow volumes for the modelled period and regulation impacts period during extreme events. During floods, all wetland models attenuated streamflow. However, during droughts, the streamflow regulation responses are variable.

Identifying which wetland model is most credible depends on whether model performance, expected streamflow regulation roles are performed or the extent of hydrological realism is prioritised. Unfortunately, neither of the wetland models considered in this comparison study were able to be acceptable for all three factors.

The third objective of the study was to compare the model suitability observed in objective 1 compares with the model performance and streamflow regulation simulated in objective 2. This was addressed in the discussion of Chapter 3. Both the qualitative and quantitative approaches to comparing wetland models showed that wetland representation, including the simulation of streamflow regulation associated with unchannelled valley-bottom wetlands, was achievable by the ACRU models. However, both ACRU and WRSM-Pitman wetland models require further validation regarding the wetlands water use.

## 5.2. Implications and recommendations for future research

From the findings of this study, a few contributions were made to the field of wetland modelling and simulating water yields from catchments with unchannelled valley-bottom wetlands (or wetlands in general). As a general rule: it appears that these results motivate for wetland representation improvements when the simulated wetland closely describes the physical wetland's characteristics and processes. This was especially reliant on differentiated wetland storage units into a soil profile and groundwater, temporally variable evapotranspiration and the amount of water available for evapotranspiration. Therefore, a best practice approach may be to prioritise a tool or wetland model with many similarities, the ability to represent the physical wetland's features and flow pathways, and physically-based and -meaningful processes and parameters (as opposed to conceptual parameters and processes). To improve daily simulations, it may be worth exploring and allowing different parameters for the ACRU wetland models compared to this study where parameters were kept the same for both models with only the wetland component and inflows pathways changing.

In terms of prioritising wetlands in the model selection process, this study demonstrated an efficient and versatile method for comparing wetlands with and without modelling which can be applied to other tools, models and physical wetland types. This is a significant benefit to wetland-inclusive modelling since there are many tools available for use, several types of wetlands and pressure to use legacy models even if, or without checking, if the wetland model is hydrologically sound. For potential users in academic and industry settings with limited time and many catchments to model, the method for comparing wetlands without modelling is repeatable and less time intensive than modelling.

Furthermore, the study provides some cautionary warnings to modellers on the potential for different water yields and streamflow regulation abilities embedded in the models. Different model outputs could lead to very different water availability, use and wetland conservation measures. This highlights the importance of contextualising the model output by identifying the wetland's impact on streamflow before applying the model to impact assessments and water resource decisions.

It would be advisable to consider the results from this study within the context of the investigation. The study was conducted within limitations related to scope and model design. In the comparison without modelling, it is yet to be determined whether the criteria will hold for other types of wetlands or if other criteria need to be added. Considering the validation, observed data sets were short and available for a few variables. Simulating wetlands relies on a water balance and confirming streamflow regulation from a wetland requires long-term streamflow monitoring over a range of various flow conditions. Considering the scope of the study, the findings were for one type of wetland and modelling was completed on a subset of the selected modelling tools. Considering the model configurations, extremes of flow pathways and parameters were assessed. For example, the previously mentioned flow pathway differences between the wetland and riparian one HRUs or in the case of WRSM-Pitman, all excess wetland storage leaving the wetland in the next time step. In terms of the wetland and catchment properties, changing wetland area or vegetation properties in response to alien vegetation invasions and clearing, erosion and disturbances from floods were not considered. Neither were the land use changes in the surrounding catchment over time accounted for, rather recent mapping was assumed to be representative and consistent for the simulation period. At best, this study serves as motivation to investigate a wetland models suitability and predictions of streamflow regulation for other case study wetlands, and for investments into strategic and long-term wetland monitoring.

For future research, addressing the limitations of this study is one way of carrying the research forward. Firstly, the study highlighted the need for strategic monitoring of streamflow, wetland storage, inflows and outflows to support model development and performance assessments. Secondly, the assessment of wetland suitability and the intercomparison from modelling needs to be tried on the other wetland types in the HGM classification and for other tools. In the comparison without modelling, there is a need to determine whether the criteria will hold for other wetland types or if there are other wetland features or criteria to be added. Additionally, it would be interesting to model the case study catchment in the tools which were not modelled (e.g. MIKE SHE coupled with MIKE Hydro River and SWAT). Would the model suitability still be an initial indicator of good model performance and the simulation of streamflow regulation for the highly suitable models in the coupled tool? Would model performance be unacceptable from SWAT? Would any of these two tools simulate the streamflow regulation expected from an unchannelled valley-bottom wetland? Would the fully-distributed model, MIKE outperform the semi-distributed models with ACRU's wetlands, proving physically-based and fully-distributed models as superior wetland model options or that increasing model performance with increasing model complexity is limited? These are all outstanding research questions. Furthermore, more models of the case study wetland with intermediate setups, flow pathways and parameters could be developed. Thirdly, phased modelling could be attempted. In this case, the simulation would be separated into periods of similar land uses and wetland properties.

In addition to increasing the modelling scope, recommendations for future research can be found in the results which were not explored or fully interpreted. For example, the different wetland storage responses from the long-term annual average and event-based water balance for the wetland or the distribution of flows in the catchment water balances across the models.

Lastly, this study highlighted an outstanding need in modelling practices. A definitive standard for a credible model for catchments with wetlands is yet to be established and agreed on. It was evident that a model with moderate model performance (good catchment streamflow but underestimated wetland evapotranspiration) could simulate the expected streamflow regulation roles expected from an unchannelled valley-bottom wetland (cf. the riparian zone HRU). This conundrum suggests that modellers may need to decide on the standards for establishing a credible model for catchments with wetlands: either the model performance, streamflow regulation or both.

## 5.3. Closing summary

This research compared wetland models and the subsequent streamflow regulation from different modelling tools and models. Setting up an unchannelled valley-bottom wetland in five models, three of which were modelled, revealed that wetland models are different and reflect the model design inherited from the software and setup choices from the modeller. Despite best efforts to setup the models as realistically as possible, the wetland models are not perfect representations of unchannelled valley-bottom wetlands. The wetland models are equipped for modelling the wetlands at monthly time-scales but show limitations for capturing the catchment hydrology at a daily time step. As a result, the suitability's of simulated wetland options for the case study wetland were variable. Similarly, the predicted water yields for the catchment over the whole simulation period and during droughts vary, while floods are always attenuated but with variable extents. This study highlighted the need for evaluating wetland characteristics and process relative to a physical wetland when modelling catchments with wetlands. More importantly, the study found that each wetland model responds similarly to several droughts and floods. This suggests that model responses are unique and identifying the model behaviour during extreme events within the historical climate records is essential to contextualising model output before applying the simulations to water resource decisions or alternative scenarios of change.

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# Appendices

## Appendix 1: Cross-validation results of rainfall interpolation

Optimal values for the mean error (ME) are low and for the standardised root mean square error (RMSE) is 1. In both cross-validation assessments, the Kriging produced the lowest ME (Figure 40) and averaging the annual, summer and winter RMSE yielded 0.97 which was closest to 1 (Figure 41). The cross-validation process in ArcGIS Pro does not consider Inverse distance weighting (IDW) in the standardised RMSE test.

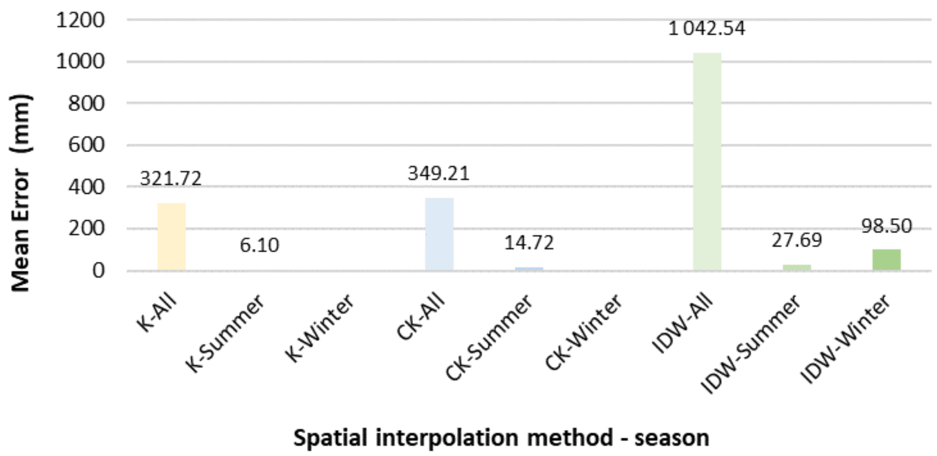


*Figure 40. Cross validation mean error from spatial interpolation of rainfall*

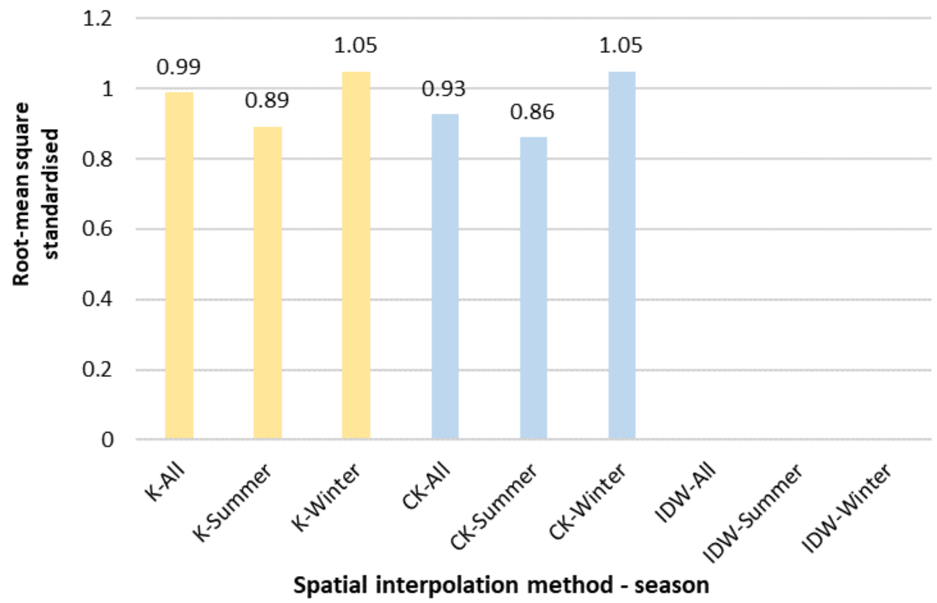


*Figure 41. Cross validation standardised RMSE from spatial interpolation of rainfall*

## Appendix 2: Evapotranspiration percentile distribution

For model performance assessments on wetland AET, Table 19 presents the percentile datasets consisting of the 10th, 20th, 30th, 40th, 50th (median), 60th, 70th, 80th and 90th percentile ET values, and the minimum and maximum ET, from the observation data and model output.

*Table 19. Wetland AET observations from remotely-sensed product (FruitLook, 2011 and updates) and three wetland models (comprehensive wetland, CW; riparian zone HRU, RZ and wetland HRU, WL)*

| Percentile | Min | 10th | 20th | 30th | 40th | 50th | 60th | 70th | 75th | 85th | 90th | Max |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Observed** | 7.83 | 19.33 | 29.31 | 33.83 | 37.65 | 49.16 | 56.21 | 71.15 | 92.05 | 112.98 | 122.25 | 187.50 |
| **CW, GROSS ET** | 28.40 | 37.20 | 43.2 | 47.2 | 54.28 | 77.75 | 98.74 | 124.8 | 128.07 | 138.18 | 138.40 | 161.60 |
| **RZ, AET** | 4.00 | 11.27 | 13.07 | 16.54 | 20.53 | 26.67 | 31.21 | 46.07 | 53.67 | 79.67 | 84.07 | 135.34 |
| **WL, AET** | 14.76 | 24.85 | 36.73 | 42.65 | 46.1 | 51.62 | 58.2 | 81.58 | 84.48 | 120.93 | 126.02 | 172.03 |