Public Comment
North Coast Restoration Policy
Deadline: 4/14/15 by 12:00 noon

RUSSIAN RIVERKEEPER®

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SWRCB Clerk

April 10, 2015

Chair Felicia Marcus and Board Members
c/o Jeanine Townsend, Clerk to the Board
State Water Resources Control Board
1001 I Street, 24th Floor
Sacramento, CA 95814

Re: North Coast Restoration Policy – NO Bank Stabilization with rock on Incised Streams!

Dear Chair Marcus and Members of the Board,

I am writing to you today regarding the proposed "R1-2015-0001" Policy in Support of Restoration in the North Coast Region (Restoration Policy) to suggest a small fix to a big problem that I have discovered within the Restoration Policy.

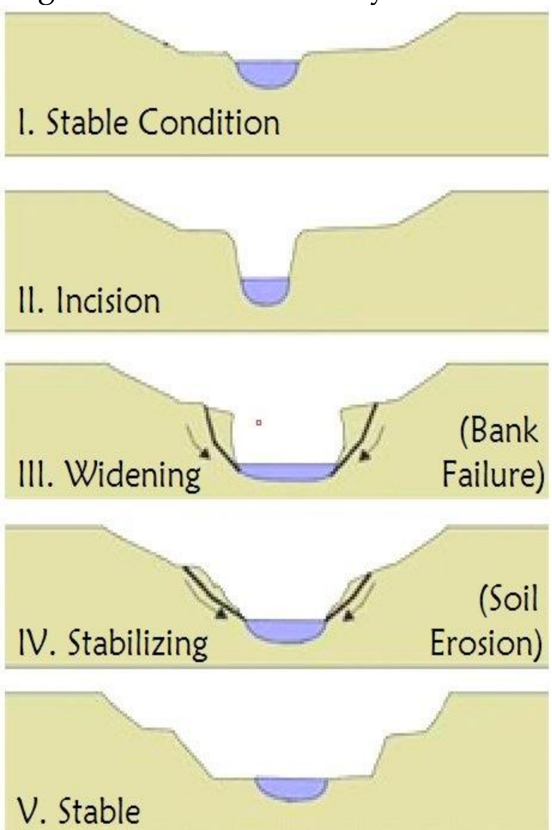
Riverkeeper Was Unable to Comment During RB1 Policy Adoption
Riverkeeper does not have the staff to comment on each and every item under consideration at the North Coast Waterboard and regrettably was not able to comment during the comment period or attend the adoption hearing in January of this year. We had to choose between commenting on what looked like a good policy or working to submit a DROPS grant to DFA and submit comments on two very bad development projects. Of course, the devil is often in the details and after carefully reviewing the Restoration Policy we feel we need to submit comments on one aspect and support everything else in the Restoration Policy. To be fair to North Coast staff we want to mention that it was our inability to comment not Region One Staff or Board ignoring our concerns that caused us to bring them to the State Board.

Incised Channelized Rivers Should NOT be Subject to Bank Stabilization with Rock
We strongly oppose the continued stabilization of river-banks in the deeply incised and channelized Russian River. Decades of gravel mining and farmers clearing land near the river as they were taught have turned ¾ mile wide meander belts into a narrow deep ditch that efficiently speeds water to the Ocean. As this Policy will guide permitting of projects and permit coordination programs, we are gravely concerned that we will continue the bad practice of tightening the strait jacket the Russian River is currently in with more bank stabilization projects on a fast permit track. We do not oppose the use of rock for stabilization for critical, immoveable public infrastructure projects but we do not believe that these types of projects are the subject of this permit.

The incised channelized state of the Russian River has lead to:

- Loss of groundwater storage as a deeper thalweg drains adjacent alluvial aquifers
- Increase in erosion rates as banks moved from shallow angles to vertical

- Complete loss of riparian vegetation in minor erosion due to cutting of former riparian forests leaving nothing to defend against erosion except shallow rooted grape vines
- Loss of majority of shallow water habitat that is critical to supporting historic salmon runs due to bank projects steepening banks – See NOAA NW Science Center paper
- Mainstem channel is a "biologic bowling alley" as energy of river is trapped in narrow channel and scours the top 4-5 ft of riverbed
- Loss of spawning gravels & marcoinvertabrates due to high scour rates related to higher energy in the river system
- Reduction in groundwater recharge rates – loss of bank permeability
- Perpetuating climate change vulnerability to more intense floods
- Rock and bank stabilization fights a streams geomorphic desires and almost always fails over time as imposing static solutions on a dynamic river is contrary to current science



- Figure 1. Graphic depicting a river cross-section as it succumbs to incision and subsequently adapts. (2003, USGS)

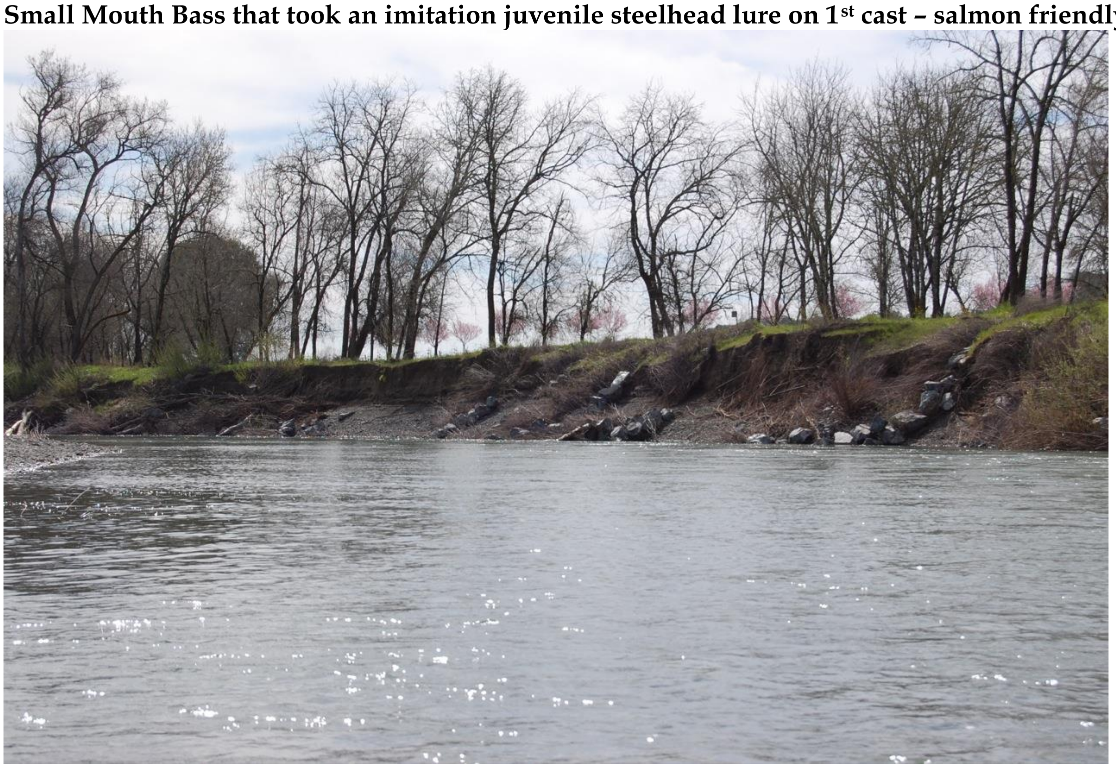
As you can see from the USGS graphic above, incised rivers evolve after initial incision and go through a widening phase before they can become stable and less erosive. What is happening today throughout the North Coast Watersheds is that we are permitting and funding bank stabilization with large rock attempting to control the river's effort to widen and stabilize itself. As we do this, we are also reducing groundwater recharge as shown by the research paper attached (setting back levees on the Consumnes River). By stabilizing banks, we are not just fighting the rivers geomorphology but we are fighting it's attempt to recharge groundwater which is of far greater value than halting the movement of sediment from temporary bank storage back into the active channel.

Bioengineering with Rock is Not Stopping Sedimentation

The pictures below show a Bioengineered Bank Stabilization Project at Stuhlmuller Vineyards on the mainstem of the Russian River. The first picture shows the project after completion in August of 2006 showing a series of rock "vanes" with vegetation planted between the vanes. This design is employed because agencies will not permit all rock rip-rap covering eroding banks recognizing it fails when flood flows weaken underlying sand and gravel and rocks slide into the river and erosion continues. The sad joke on both taxpayers-who pay 50% or more-and the river ecosystem is that these fail in every single case except one: where bedrock substrate was present in the lower river environments. The second picture was upon completion. Russian Riverkeeper Staff went diving and fishing to see what species were using the "habitat" that was restored and it was 100% pike minnow and small mouth bass, including one we caught immediately when we used an imitation steelhead fry lure. The third and fourth pictures are on March 2008 after a modest 21,000 cfs (HEA) peak flow event in January of 2008 where a massive amount of sediment eroded from the project site – this wasn't supposed to occur after we spent $1,500,000 in public taxpayer funds! Today the site is almost 100% rock after the two other fixes failed leaving it in worse shape than when they started the project. This is not restoration (nor should it be considered as such) but the fact that it is funded by DF&W is disturbing.

Stuhlmuller Bank Project August 2006, being completed

Small Mouth Bass that took an imitation juvenile steelhead lure on 1st cast – salmon friendly??



March 2008 after modest 5-year flow event – massive loss of sediment despite BA work.

Additional views in March 2008 showing large loss of sediment – This is private property protection!

Current Science and Research Does Not Support the Hard Armoring of Banks.
Several recently published studies have looked at current and future climate scenarios and the associated effects that more intense flooding events and drier, hotter summers will have upon our watersheds (see 5 studies attached). These studies all have a common theme-continued reliance upon bank stabilization projects will result in the disconnection of natural floodplains effectively diminishing hydrologic connectivity and degrading the biological integrity and functionality of ecosystem services. Please review each of the research papers attached before you make the decision to approve the "R1-2015-0001" Policy in Support of Restoration in the North Coast Region. By reviewing the current science, you will clearly see that the hard armoring of banks using rock should be an extremely limited practice and only used in conjunction with the protection of critical public infrastructure projects.

Suggested Remedies to Our Concerns over Use of Rock in Bank Projects
What do we recommend changing? Here are some options:
1. Add the phrase, "with no rock" after any use of the term stabilization
2. Call out the issue of appropriate solutions on incised channelized streams and the need to balance bank stabilization with groundwater recharge, flood capacity and pollutant attenuation and that in many cases bank stabilization can reduce the level and value of ecosystem services that will be critical due to predicted climate change impacts.
3. Remove any mention of bank stabilization from this Policy.

Please revise this policy to end the promotion of projects that damage the river's ability to support beneficial uses and remove bank stabilization from consideration in this Policy in Support of Restoration.

Thank you for consideration of our comments.

Sincerely,

Don McEnhill
Executive Director

Attachments:
1) Modeling study of groundwater and surface water interaction using high resolution integrated model by Yunjie Liu
2) Combined Effects of Climate Change and Bank Stabilization on Shallow Water Habitats of Chinook Salmon by Jeffrey C. Jorgensen et. all
3) Bank Erosion as a Desirable Attribute of Rivers Joan L. Florsheim et.all
4) Russian Riverkeeper Publication
5) Technical Memorandum CDF&W

Modeling study of groundwater and surface water interaction using high resolution integrated model

YUNJIE LIU

B.S. (China Agricultural University) 2010

THESIS

Submitted in partial satisfaction of the requirement for the degree of

MASTER OF SCIENCE

in

Hydrologic Sciences

in the

OFFICE OF GRADUATE STUDIES

of the

University of California, Davis

Approved

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Dr. Graham Fogg, Chair

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Dr. Thomas Harter

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Dr. Reed Maxwell

Committee in Charge

September 2014

Yunjie Liu

September 2014

Hydrologic Sciences

Abstract

The groundwater and surface water comprise a single source of water resources. Efficient and sustainable water resource management requires using groundwater and surface water conjunctively. Worldwide, many water shortage problems come from the fact that neither the timing nor the location of precipitation coincide with water demands. Climate change makes this problem even worse. For California particularly, the warming trend is shifting more precipitation to fall as rain rather than snow during winter season, thereby reducing snow pack in Sierra Nevada Mountains. In addition, snowmelt is occurring earlier in spring due to warmer temperatures, therefore reducing the availability of snowmelt water that contributes to stream flow and surface reservoirs during the dry summer season. Climate projections also suggest that winter floods will become more frequent, as will hotter and drier summers. The imbalance in time of water distribution within a year (wet, dry season), and between years (wet, dry year), as well as extreme climate events (for example, 1997 flood, 2012-2014 mega drought for California), create great challenges for water resource management. It is especially true when climate change effect is expected to continue.

This study evaluates winter floodplain inundation as a strategy of capturing and storing excess winter flood water beneath Central Valley floor to restore groundwater for local subsurface reservoir development. The parallel, variably saturated flow modeling code, ParFlow, is chosen to model the spatial and temporal patterns of surface water and groundwater interaction in heterogeneous subsurface under floodplain inundation at lower Cosumnes River floodplain. Particularly, the mechanics of groundwater and surface interaction in heterogeneous subsurface is investigated. Development of local subsurface groundwater reservoir on water resource management is discussed.

Results of this study show that groundwater and surface water interaction under floodplain inundation is controlled by the heterogeneity of subsurface, primarily the connectivity of heterogeneity, as well as flood water inundating dynamics. A regional, subsurface reservoir can be augmented through floodplain inundation practice. However, its role of mitigating climate change impact on water resource management on a long time frame needs further investigation.

Quotes

"When the well is dry, we know the worth of water"

----Benjamin Franklin

"All models are wrong, but some are useful"

----Gorge E. P. Box

"*Make things as simple as possible, but not simpler*"

----Albert Einstein

"*越十年生聚，而十年教训。二十年之外，吴岂为沼乎？*"

----伍子胥

Acknowledgements

Today is May 11st 2014, Mother's day. I especially thank my parents, who consistently support me through my graduate studies. Their support is particularly special and important to me to come here studying at Davis.

I would like to thank my advisor Dr. Graham E. Fogg for the opportunity to work on this project and for his numerous support and assistance, scientific insights and patience during my graduate studies. I also would like to thank other members of my thesis committee, Dr. Thomas Harter and Dr. Reed Maxwell for their effort of advising and commenting my thesis. And I would like to thank funding from The Nature Conservancy for making this project possible.

Thanks for colleagues and friends in the "Hydrology Think Tank A" lab, Nicholas Newcomb, Charlie Paradise, Dylan Boyle, Rashid Owlia, Jeannette Sager, Nicholas Engdahl, Peter Dennehy, Emily Hathaway and so on for making the lab atmosphere so relaxing and fun. Thanks for Dr. Wesley Christensen and Dr. Jim Trask for reading and commenting my thesis.

Many other friends have accompanied me through my time at UC Davis, teaching and helping me in various ways. Thank you all my friends at Davis, Olin Applegate, Francis Corbett, Shijian Zhuang, Anda Zheng, Sailin Xia, Ni Zhang, Xiao Qiu, Man Zhang and the group of people I have always been hanging out with. You guys made the experience totally memorable and enjoyable.

Table of Contents

Chapter 1 Introduction

1.1 Overview of groundwater and surface water interaction

Groundwater and surface water are interconnected. The development and contamination of one will affect the other, because groundwater interacts with surface water in almost all types of landscapes (Winter *et al.* 1998, Winter 1999). The exchange between groundwater and surface water has controlling influence on stream chemistry, nutrient flux and near river biota (Cey *et al.* 1998, Packman 2003, Niswonger 2006, Niswonger and Fogg 2008). Understanding the exchange of water, energy and biogeochemical matter between surface water and groundwater is critical for addressing the water quality and quantity problems as well as for maintaining the health of the ecosystem (Winter *et al.* 1998). The interaction between groundwater and surface water, however, can depend on multiple factors, including surface topography, stream channel geometry, underling geologic structure, subsurface hydraulic parameters, temporal variation in precipitation and local groundwater conditions (Cey *et al.* 1998, Woessner 2000, Niswonger 2006, Niswonger and Fogg 2008). In the case where the water table is sufficiently shallow to form a fully saturated connection between the surface water and groundwater, the surface water and groundwater are viewed as hydraulically connected. The spatial and temporal variability of groundwater and surface water interaction is dominated by saturated flow, which relies on the head difference between surface water and beneath groundwater, and the hydraulic conductivity of sediments between them (Sophocleous 2002, Fleckenstein 2004, Fleckenstein *et al.* 2006, Frei 2008, Frei *et al.* 2009, Brunner *et al.* 2010). In the case of a deeper water table, unsaturated zone exists beneath surface water body. The surface water and groundwater are conventionally viewed as hydraulically disconnected in this case, thus the interaction between them are dominated by variably saturated flow and commonly assumed to be one direction only, where surface water seeps vertically to water table, and further drawdown of the water table does not increase the rate of stream flow loss to the groundwater (Winter *et al.* 1998, Fleckenstein 2004, Fleckenstein *et al.* 2006, Niswonger 2006, Niswonger *et al.* 2008, Frei *et al.* 2009, Brunner *et al.* 2010). In this process surface water must travel through unsaturated zone downward before it can interplay with groundwater (Fleckenstein 2004, Fleckenstein *et al.* 2006, Birdsell *et al.* 2005, Vazquea-Sune *et al.* 2007). The variably saturated flow process and aquifer response is a function of surface water depth, vadose zone thickness, aquifer hydraulic conductivity, and heterogeneous structure of subsurface geology (Fleckenstein 2004, Frei 2009, Shanafield 2012, Sager 2012), which results in a complex, spatial and time scale dependent system. The unsaturated zone residence time can be hours to years, and even longer (Birdsell *et al.* 2005, Sager 2012). Therefore, the common numeric assumption of constant and instantaneous transfer from surface infiltration to groundwater recharge is not valid in this case (Brunner *et al.* 2010). The groundwater surface water interaction process is a dynamic process, meaning the hydraulicly connected and disconnected states between groundwater and surface water are not static. Factors such as, stream flow, infiltration, pumping, subsurface heterogeneity, and vegetation can cause a connected state to be disconnected and reversely at various temporal and spatial scales, which results in distinct flow patterns (Fleckenstein 2004, Fleckenstein *et al.* 2006, Niswonger 2006, Niswonger and Fogg 2008, Frei *et al.* 2009, Sager 2012, Desilets *et al.* 2008, Brunner *et al.* 2010).

The so called "hydraulically disconnected" condition is commonly found in arid and semiarid regions, where water table is typically well below the surface water body. It is also often observed at regions where intensive groundwater pumping dramatically lowers the water table. The Central Valley, California, USA serves as a good example of the latter, where the water table has been lowered due to decades of groundwater pumping for agricultural and municipal use. This study focuses at the north part of the Central Valley (Sacramento County) where two major cones of depression were developed on the north and south of Cosumnes River due to intensive pumping. Considering that the groundwater pumping will not likely be decreasing due to increasing water demand for agriculture and urban uses worldwide in the future; and the fact that arid and semiarid regions occupy roughly one third of the Earth's land surface and are expected to grow due to climate change (Schlesinger 1990), the disconnected surface water and groundwater condition will be more common in the future (Desilets *et al.* 2008). Thus understanding how the interaction between groundwater and surface water happens and what factors control this process, as well as its role of supporting health eco-systems (Woessner 2000) would benefit multiple disciplines such as environmental conservation, conjunctive water resource management, contamination control and climate change studies.

1.2 Subsurface connectivity of heterogeneity and surface water and groundwater interaction

It has long been recognized by the hydrogeology community that the material properties of the subsurface are highly variable in space due to the spatial and temporal complexity of geologic processes (Neuman 1982, Fogg 1986, Kolterman and Gorelick 1996, de Marsily *et al.* 1998). In the past 50 years, numerous efforts have been devoted to conceptualizing and modeling the subsurface heterogeneity. Renard *et al.* (2011) categorized these work into three phases which are summarized below. In the first phase, most models considered an ensemble of regions having constant equivalent properties. In the second phase, small scale variability has been considered as a key feature, and geostatistical methods, such as uni-variant Gaussian distribution, multi-variant Gaussian distribution, were intensively used to model the subsurface spatial variability. At the beginning of the third phase, the connectivity of heterogeneity is considered a property that strongly affects the spatial pattern of the hydraulic parameters, thus consequently affects groundwater flow and solute transport (Fogg 1986, Weissmann and Fogg 1999, LaBolle and Fogg 2001, Western *et al.* 2001, de Marsily *et al.* 2005, Fleckenstein *et al.* 2006, Niswonger and Fogg 2008, Enghdal *et al.* 2010, Vogel *et al.* 2010, Renard *et al.* 2011, Sager 2012).

The connectivity is referred to which specific hydro-facies (sediment types) are connected in a spatial distribution, and reflects structural heterogeneity (Lee *et al.* 2007). More broadly, the connectivity denotes the extent to which the connected features, such as sands and gravels, occur in a hydrologically relevant spatial pattern (Western *et al.* 2001). A good example in hydrogeology is the preferential flow path ways: connected higher permeability sediments; and the flow barrier (aquitard) or confining bed: connected low permeability sediments.

Fogg (1986) concluded that the groundwater flow and solute transport can be controlled more by the connectivity of high permeable sediments (sand and gravel body) than the hydraulic conductivity of the sediment itself. The flow and transport process can be completely different with the presence of well-connected higher permeability sediments. Many recent research on flow and transport do support this conclusion (LaBolle and Fogg 2001, Zappa *et al.* 2006, Vassena *et al.* 2010). Failing to capture connectivity of heterogeneity can introduce bias on groundwater flow and solute transport modeling (Gomez-Hernandez and Wen 1998, Western *et al.* 2001, Zinn and Harvey, 2003, Lee *et al.* 2007), while inadequate characterization of connectivity can largely change the equivalent hydraulic property of subsurface and affect the overall system behavior (Renard *et al.* 2011). Groundwater flow modeling conducted by Lee *et al.* (2007) on two geological models that share similar statistics but with different connectivity structures show that the pumping-drawdown relation in the model with higher degree of lateral connectivity of high permeable sediments generated by Transition Probability Geostatisics (TPROGS) fits better with observed data than the pumping drawdown relation simulated in the random heterogeneous model generated by sequential Gaussian simulation (SGS). Solute transport modeling in highly connected geologic model shows earlier peak breakthrough and extensive tailing comparing to solute transport in less connected model. The connectivity of heterogeneity is the dominant cause of so-called anomalous transport behavior (Willman *et al.* 2008, Bianchi *et al.* 2011). In a modeling study on MADE site, Bianchi *et al.* (2011) conducted particle tracking simulation in three heterogeneity geological models generated by Sequential Gaussian Simulation, Sequential Indicator Simulation and Transition Probability Geostatisitics respectively. They found that, in all three simulations, the fraction of particle paths within high permeability connected channel accounts for about half of particle paths, though the volumetric fraction of connected channels is small. As a result, the simulated break through curves show sharp peaks at early times and extensive tailing.

The groundwater head variation, which ultimately drives groundwater flow, is less sensitive to hydraulic conductivity fluctuation in three dimensional models but more sensitive to the connectivity of the high permeable zones as compared to two dimensional models (Fogg 1986). This is because the number of high permeable pathways that fluid can bypass the low permeable sediments increases in three dimensions, therefore the head drop can be smaller (Fogg 1986). However, the variation in head seems not necessarily sensitive to the degree of connectivity of heterogeneous in three dimensional models (Fogg 1986). This indicates that limited head measurements in field may not be sufficient to locate and identify the connectivity of high permeable units (Fogg 1986, Renard *et al.* 2011). Other research, however, shows that the head variance, to some extent, can be used as an indicator of subsurface connectivity of heterogeneity: higher head variance indicates poor connectivity (Frippiat *et al.* 2009). In a study, Giambasiani *et al.* (2009) plotted one year groundwater head fluctuation at Namoi catchment, Australia onto three dimensions, and found that the head variation indicates the pathways of pressure transfer that coincides with Namoi River paleochannels. They concluded that head measurements can be used to indicate subsurface geometry and delineate the location of aquifers. Information from other variables such as water temperature and pH can be very beneficial in delineating aquifers together with the head data. The accuracy and associated uncertainty of many modeling applications are largely affected by the conceptual subsurface model structure (Nilsson *et al.* 2007) and the degree of subsurface heterogeneity characterizations (Engdahl *et al.* 2010, Bonomi 2009). Therefore, having other sources of information such as prior geological knowledge, geophysical observations or tracer test observations as conditioning information to infer subsurface connectivity is critically important in developing plausible geological models (Fogg 1986, Kerrou *et al.* 2008, Fernandez-Garcia *et al.* 2010, Renard *et al.* 2011).

Though in the past, many researches have shown the importance of connectivity of heterogeneity on groundwater flow and transport modeling, it is hard to describe quantitatively. Knudby and Carrera (2005), Knudby and Carrera (2006), Knudby *et al.* (2006) recently analyzed several connectivity measurements including the statistic connectivity, flow connectivity and transport connectivity, the ratio of effective permeability to the geometric mean of the permeability field as well as hydraulic diffusivity as indicator of connectivity. Western *et al.* (2001) used connectivity function to characterize the spatial connectedness. Xu *et al.* (2006) introduced connectivity index in describing the connectedness of fractures in rock networks. Renard and Allard (2013) introduced the concept of static connectivity matrices which relies only on the connectivity of the subsurface geological structure and dynamic connectivity matrices that related to the specific physical process. Discussion of quantifying the connectedness is beyond the scope of this thesis.

During the last decade, much research has been done on the topic of the impact of structural heterogeneity, including connectivity, on hydrological processes both in surface hydrology (Pringle 2001, Pringle 2003, Ocampo *et al.* 2006, Tetzlaff *et al.* 2007, Bracken and Croke 2007, Mueller *et al.* 2007, Meerkert *et al.* 2009, Appels *et al.* 2011) and subsurface hydrology (Borgne *et al.* 2006, Knudby *et al.* 2006, Zappa *et al.* 2006, Schaap *et al.* 2008, Lehmann *et al.* 2008, Morin *et al.* 2010, Fernandez-Garcia *et al.* 2010, Fiori *et al.* 2010, Ali and Roy 2010). However, as a critical component in hydrological cycle, the groundwater and surface water interaction in heterogeneous subsurface has not been investigated enough. Several studies have pointed out the essential role of structure heterogeneity on controlling flow and interaction process, as well as the chemical, biological process associated with it (Oxtebee and Novakowski 2002, Niswonger 2006, Niswonger and Fogg 2008, Fleckenstein *et al.* 2006, Frei *et al.* 2009, Engdahl *et al.* 2010, Sager2012, Irvine *et al.* 2011, Pulido-Velazquez *et al.* 2011, Moffett *et al.* 2012, Giambastiani *et al.* 2012).

Studies on small river reach scale suggest that the variability of groundwater and surface water exchange in space is dominated by interface heterogeneity (Wagner and Bretschko 2002, Cardenas *et al.* 2004, Salehin *et al.* 2004), and the connectivity from surface to subsurface. That is the preferential flow path (connected high permeable unit) and the flow barrier (low permeable unit) (Frei *et al.* 2009, Sager 2012, Niswonger and Fogg 2008). Moreover, the variation can occur at a very small scale of meters to centimeters (Woessner 2000, Conant 2004). In a field study, Conant (2004) mapped streambed temperature at various locations at a certain depth to investigate stream groundwater interaction at a 60 meters river reach. The results show high spatial variability in recharge zones that only 5 to 7 percent of the area accounts for 20 to 24 percent of total recharges for the river reach. Similar approach was used in a study by Schmidt *et al.* (2006) on investigating groundwater surface water interaction at a manmade stream. They estimated the flux exchange by applying analytical solution to the heat conduction-advection equation on measured vertical temperature profile and found that two high recharge zones account for 50 percent of total recharge on only 20 percent of the river reach. In a soil column drainage lab study, Schaap *et al.* (2008) applied neutron tomography on mapping three dimensional water content profile evolution in a well-defined artificial heterogeneous column. Their results show that the water content profile evolution is greatly influenced by the connectivity of the coarse sand: water content changes faster in coarse sand connected channels than in unconnected coarse sand. A field study conducted by Oxtebee and Novakowski (2002) on fracture aquifer system at twenty mile creek shows that the groundwater and surface water exchange in the vicinity of the creek is limited due to the extremely poor vertical connections between fractures. In a modeling study on stream aquifer interaction at lower Cosumnes River, California, Frei (2008), Frei *et al.* (2009) obtained similar results that over 90 percent of stream recharge happen in the preferential flow. Another modeling study at the same study site conducted by Sager (2012) showed that the preferential flow path is filled up within hours to days, creating localized hydraulic connectivity from surface water to groundwater. Subsequently, water slowly soaks into the relatively low permeable sediments near by the connected high permeable units. However, due to the limited volume of the preferential flow path, its storage capacity is comparatively small and may not play an important role on groundwater storage change in long term (Sager 2012). These features are particularly important for groundwater restoration projects such as artificial recharge practices.

The connected low permeable lens, the so called flow barrier, can be as important as the preferential flow path in shaping groundwater and surface water interaction process. As shown in several studies, the perched aquifer can be formed where focused infiltration is obstructed from flowing downward by low permeable flow barrier (Niswonger 2006, Niswonger and Fogg 2008). The perched aquifer can significantly reduce the seepage rate from surface into subsurface and redistribute water laterally, even generate base flow to stream when the flow in stream receded in dry season. In a study on the entire Rocky River catchment, where the river overlies on a layer of unconsolidated sandy clay and clayed sand that contains minor gravel lens, Banks *et al.* (2011) showed that the dominant groundwater source to the stream is from the shallow perched aquifer system in the catchment headwaters. The perched aquifer provides base flow down the length of Rocky River and maintains flow for a long period of time. It is also because of the contribution of shallow perched aquifer that the fresh water system in river is maintained in otherwise saline regional fractured rock groundwater system, which highlights the importance of perched aquifers on controlling the water quality of stream flows (Banks *et al.* 2011). The rate of perched aquifer discharge to the stream and the duration of discharge are controlled by the hydraulic conductivity of sediments surrounding the stream, hydraulic conductivity of the perched zone, the depth of stream penetration, the size of perched zone as well as the riparian vegetation (Niswonger and Fogg 2008, Carter *et al.* 2011).

River basin scale is the traditional focus of groundwater and surface water interaction studies, where the water budget is the main concern. In a regional numerical model, the hydraulic parameters are often assumed to be homogeneous over a large river reach, thus the interface of surface water and groundwater may be represented only approximately (Frei *et al.* 2009). A few regional scale groundwater and surface water interaction studies have addressed the effect of subsurface heterogeneity. In a study at the Cosumnes river basin, Fleckenstein (2004), Fleckenstein *et al.* (2006) found that during moderate and high flow conditions, half of the total seepage along the river happens on only 10 to 15 percent of river channels where there is connected high permeable preferential flow path from surface to subsurface. Comparatively, in homogeneous model, over 30 percent of the river channel can contribute the same amount of seepage under the same flow conditions. As Kalbus *et al.* (2009) concluded: the well connected high-permeable zones can lead to very high fluxes concentrated in small area. These localized seepage spots can create hydraulic connection between surface water and groundwater in thick vadose zone and can reverse the hydraulic gradient, thereby turning some river reaches from losing to gaining at particular period of a hydrologic year (Hathaway *et al.* 2002, Fleckenstein 2004, Fleckenstein *et al.* 2006). Simply referring a river reach to be losing or gaining is not proper because the groundwater and surface water interactions can lead to gradient reversing between seasons (Hinkle *et al.* 2001, Krause *et al.* 2007). Moreover, although the regional water table may be sufficiently below the stream stage to seem consistent with the standard assumption of a disconnected stream-aquifer system (i.e. further reductions in the regional water table do not induce further increases in stream flow losses), these local connections that can become saturated all the way up to the streambed can create the two-way, groundwater and surface water connections, rendering invalid, the common "disconnected" assumption.

On the ecological aspect, groundwater is a key component of environmental flows (Sophocleous 2007). Maintaining or restoring a health river ecological system, or more broadly the groundwater dependent ecosystem, has been the focus of many groundwater and surface water interaction studies. Protection of river ecological system is an important task in sustainable groundwater resource management practice (Barron *et al.* 2012). The riparian ecosystems serve as the ultimate expression of groundwater and surface water interactions (Webb and Leak 2006) and could serve as an important indicator of the health of the river ecosystem. Riparian vegetation is sensitive to water table variations. Small declines in the water table can cause extirpation of riparian species (Stromberg *et al.* 1996), particularly in dry seasons when river runoff diminishes and the water table drops. The perched aquifer system, however, can create local shallow groundwater system under favorable hydro-geological conditions, which could support riparian vegetation and possibly sustain some stream base flow in dry season (Niswonger 2006, Niswonger and Fogg 2008). Yet, due to the limited storage capacity of perched aquifer systems, it may not be able to maintain enough base flow to streams that fish and other organism lives on. But still, it can enhance the lateral and upward hydraulic gradient and provide based flow to a stream, therefore provide necessary conditions for supporting hyporheic creatures and riparian vegetation (Niswonger, 2006; Niswonger and Fogg, 2008). In a study, Rains *et al.* (2006) showed that the perched aquifer can actually sustain spring ponds by contributing around 50 percent of the inflow to vernal pools. Under very low flow conditions, the little contribution of base flow from perched aquifer is crucial in determining the hydro-chemical conditions and resulting ecological stress during a period that may coincide with the main vegetation growth period (Krause *et al.* 2007). In their study on Havel River, Krause *et al.* (2007) concluded that groundwater contribution accounts for only 1 percent of the annual total river discharge, but the ratio goes up to 10 percent during low flow conditions.

With respect to biogeochemical aspects, the preferential pathway and the perched system create totally different residence time scale that water flows through unsaturated zone, and different degree of groundwater and surface water mixing. The longer residence time favors denitrification (Mastrocicco *et al.* 2013). The more mixing happens the higher possibility of attenuating and removal of pollutant before they either enter groundwater from surface water or enter surface water from groundwater (Conant *et al.* 2004). In addition, when nitrate reach perch aquifer, it can be efficiently removed by riparian vegetation (Hayashi and Rosenberry 2001).

1.3 Climate change and groundwater surface water interaction

Data and climate models show an ongoing warming trend due to the increasing concentration of greenhouse gases (Schledinger *et al.* 1990, Barnett *et al.* 2005, Bates *et al.* 2008, IPCC 2007 AR4, IPCC 2013 AR5). In fact, the decade of 2000's has been the warmest on record. The global combined land and ocean temperature has increased 0.91 degree Celsius over the period of 1901 to 2012 (IPCC 2013, AR5). The warming trend is strongly affecting many aspects of the global hydrologic cycles, such as increasing and intensifying precipitation, reducing snow packs in mountains, reducing soil moisture, impacting the groundwater availability, the surface water quality and quantity, the water demands etc. (Gleick 1989, Kundzwicz and Somlyody1997, Kiparsky and Gleick 2003, Labat *et al.* 2004, Huntington 2006, DWR 2008, Dery *et al.* 2009). These changes in hydrology, in turn, would lead to surface land cover change (Van Mantgem and Stephenson 2007). For example, reducing in soil moisture in arid/semiarid regions can cause the land cover shifting from grassland to shrubs (Schledinger *et al.* 1990); the declining of water table beneath streams would lead to vegetation shifts from woody vegetation to low lying shrubs in riparian environment (Stormberg *et al.* 1996); water table declining can also result in the disappearance of wetland systems (Kiparsky and Gleick 2003). Land cover and eco-system change as well as changes in hydrologic patterns, such as rain fall distribution and evapotranspiration, can strongly affect groundwater surface water interaction (Scanlon *et al.* 2006, Van Roosmalen *et al.* 2009).

The temperature increasing associated with climate change is shifting the seasonal stream flow pattern to an earlier (and shorter) spring snowmelt and increasing winter runoff in snowmelt dominate streams, such as rivers in California (Barnett *et al.,* 2005, Vicuna and Darcup 2007, DWR 2008, Huntington and Niswonger 2012). Consequently, summer flow and soil moisture would decline substantially and summer dry begins earlier and at a higher degree (Miller *et al.* 2009). The winter flow and flood event can be more frequent and at higher magnitude (Gleick and Chalecki 1999, Drogue *et al.* 2004, DWR 2008). These trends will likely result in increased chances of winter flooding, possibly increased changes of drought, and hotter, dryer summers. Such trends will in turn stress the river bank eco-systems (e.g., riparian vegetation) and aquatic species (e.g., trout and endangered fall run of Chinook salomon) that depend on streamflow, groundwater, and soil moisture to survive in dry season (Fleckenstein 2004, Fleckenstein *et al.* 2006, Niswonger 2006, Harvey *et al.* 2006, Rieman *et al.* 2007, Niswonger and Fogg 2008, Sabo and Post 2008, Tavakoli and De Smedt 2012). In addition, the warmer climate and the depleted summer flow could further have profound impact on stream and soil water temperature, which, in turn, influence stream ecology and terrestrial biogeochemical reactions (Green *et al.* 2011, Barron *et al.* 2012). Climate change directly impacts surface water by long term changes in climate variables, such as temperature, precipitation, evapotranspiration. The impact on groundwater and groundwater-surface water interactions, on the other hand, is indirect and hard to address. Our knowledge about climate change on groundwater and surface water interaction is still limited (Sophocleous 2004, Woldeamlak *et al.* 2007, Jyrkama and Sykes 2007, Green *et al.* 2011, Huntington *et al.* 2013).

Climate change effect on rainfall patterns and stream flow patterns propagate to groundwater mainly due to increased demand for groundwater stemming from less snow, and hence surface water storage. Climate change likely also affects groundwater recharge. In the study area, Cosumnes River basin, however, most of the non-stream-related groundwater recharge comes from summer irrigation, so unless agriculture changes significantly, the recharge may not change appreciably. In upland areas such as the Sierra Nevada Mountains, however, climate change may change groundwater recharge and discharge regimes substantially. Field work and modeling conducted by Huntington and Niswonger (2012) in a small watershed of the northern Tahoe Basin indicate that future warmer temperature will lead to earlier snow melting and shallower stream stage in spring, therefore less groundwater discharge to stream in spring and summer. They showed that the groundwater will likely deplete during summer time due to early drainage, resulting in less available water that can provide base flow to streams. The decrement of summer flow can be as high as 30 percent. Among precipitation change and temperature change, the impact of former on groundwater recharge is greater than the impact of latter, at least for semiarid regions. However, the change in annual groundwater recharge is higher than the corresponding annual change in precipitation, indicating amplified impact on groundwater system in response to climate change (Ng *et al.* 2010, Crosbie *et al.* 2013). The near surface shallow aquifer is the most sensitive part of groundwater in responding to climate change, because it directly exposes to precipitation and evaporation (Sulis *et al.* 2011, Ali *et al.* 2012). The deep aquifer system, on the other hand, can play an important role on buffering the warmer temperature caused summer flow depletion and shifting timing of peak stream flow and changing flow patterns (Tague *et al.* 2008, Chang and Jung 2010). Similar results were obtained by Maxwell and Kollet (2008) on a modeling study of a semiarid watershed using an integrated watershed model from energy balance perspective. They found that the groundwater storage serves as the moderator of watershed response to climate change. Notably, in the critical zone, where water table is 2 to 5 meters below land surface, there is strong correlation between water table depth and surface energy responses.

Keep in mind Global Circulation Model (GCM) projections on future climate conditions usually associate with very high uncertainties (Jasper *et al.* 2004, Christensen and Lettenmaier 2007, Goderniaux *et al.* 2009, Rossler *et al.* 2012). In addition, it may not provide adequate information on likely climate conditions because of lacking of seasonal variation and transient fluctuations on the appropriate temporal and spatial scales (Goderniaux *et al.* 2011). Moreover, the GCMs usually show agreement on temperature projection, but the precipitation projections are poorly agreed, warning us the reliability of implementing information directly from GCMs (Jasper *et al.* 2004, Allen *et al.* 2010). Furthermore, different terrain and geological conditions as well as dynamic vegetation changes can strongly modify climate change effect on groundwater, groundwater-surface water interactions and groundwater depended ecosystem (Tague *et al.* 2008, VanRoosmalen *et al.* 2009, Sulis *et al.* 2010, Barron *et al.* 2012, Green *et al.* 2011, Ali *et al.* 2012). The impact of groundwater abstraction on groundwater and surface water interaction can, sometimes, be rather more significant than the impact of climate change (Barron 2012, McCallum *et al.* 2013).

Accepting the fact that climate change is affecting the entire hydrosphere, the response time lag and magnitude of different parts of the hydrological system to these changes are different (Van Roosmalen *et al.* 2009). The groundwater response to climate fluctuations tends to occur more slowly than surface water body due to its indirect exposure to climate change. Thus, it is difficult for short time scale assessment (Sophocleous 2009, Chang and Jung 2010, Mayer and Newman 2011). The delay in groundwater system responses caused by surface water fluctuation, and reversely, can be years to decades, thus requiring long term simulation and analysis (Rassam 2011, MacCallum *et al.* 2013). Knowing the scale of response time frame, is very important for water resources management, infrastructure design and policy makings (Sophocleous 2012, Bredehoeft 2011, Wolton 2011).

1.4 Modeling Challenge

From small river reach scale to large regional scale, modeling groundwater and surface water interactions in highly heterogeneous subsurface, including variably saturated flow and exchange processes remains a computational challenge (Frei 2008, Frei *et al.* 2009). This is because of the highly non-linear aspect of partially saturated flow, three-dimensional heterogeneity requiring large systems of equations, and the often sharp contrast in permeability between aquifer and aquitard materials in most geologic systems. However, integrating surface water and groundwater modeling allows a complete analysis of the feedbacks between land surface hydrologic process and groundwater flow process, thus can provide means of evaluating the impact of land use, irrigation, climate change on both surface and groundwater resources from an integrated perspective (Sophocleous 2000). With advanced computing technology, various complex physical based integrated models were developed and applied on modeling groundwater surface water interactions. Jones *et al.* (2008) studied the groundwater and surface water interaction on a 75 km^2 watershed in Canada using InHM. Werner *et al.* (2006) modeled the groundwater and surface water interaction on an even larger catchment (420 km^2) in Australia employing MODHMS. Li *et al.* (2008) investigated the surface water and groundwater interaction in a 286 km^2 watershed in Canada on HydroGeoSphere. Kollet and Maxwell (2008) studied the stream base flow residence time on a watershed with the size 1000 km^2 at Oklahama on ParFlow. Huntington *et al.* (2012, 2013) modeled the impact of climate change on groundwater and surface water interaction at mountain watershed using GSFLOW. Sulis *et al.* (2011) investigated the impact of climate change on groundwater and surface water interaction at a medium watershed in Canada using CATHY.

This study is a follow up of previous research at Cosumnes River, California on the groundwater and surface water interactions in heterogeneous alluvial fan depositional system (Fleckenstein 2004, Niswonger 2006, Frei 2008, Meirovitz 2010, Sager 2012). The fully coupled surface water and groundwater water flow model, ParFlow, is chosen. The numeric model covers 1652 km^2 including lower Cosumnes River watershed and parts of American River watershed. The Cosumnes River, California, receives great attention in hydrology, hydrogeology and ecology communities for its uniqueness as the last, major undammed river in California. The intense groundwater pumping for agriculture and municipal use has lowered the groundwater level in Cosumnes River basin and lead to no flow condition in late summer and early fall. This creates obstacles for salmon migration and spawning. This research is aimed at understanding the spatial and temporal patterns of groundwater and surface water interaction and the controlling factors of the exchange process at regional scale. It also explores benefits of floodplain inundation on restoring groundwater and mitigating climate change impact on California water resources.

Chapter 2 Study area geology and hydrology

2.1 Study area

The California central valley is the home to some of California's most productive agriculture. It consists of the Sacramento Valley in the north and the San Joaquin Valley and Tulare Lake Basin in the south. This study focuses on Sacramento County located in the southern Sacramento Valley, with emphasis on the Cosumnes River catchment (Figure 2.1). The Sacramento County covers about 2570 square kilometers. The county extends from the lower delta on the west to foothills of Sierra Nevada Mountains on the east. Most of the county is flat and at an elevation close to sea level, with exception on the east boundary, where the elevation goes up to several hundred meters at the mountain foothills. The climate is typical Mediterranean climate with cool and rainy winter but hot and dry summer. Most of the rainfall (over 75 percent) happens during November to March (Fleckenstein 2004, Fleckenstein *et al.* 2006). Major rivers in the county include American River, Sacramento River and Cosumnes River. The numeric model covers the lower Cosumnes River basin and parts of American River basin and Sacramento River basin, spreading 1652 km^2. One can refer to Meirovitz (2010) for details about the selection of the boundaries of numeric model domain.

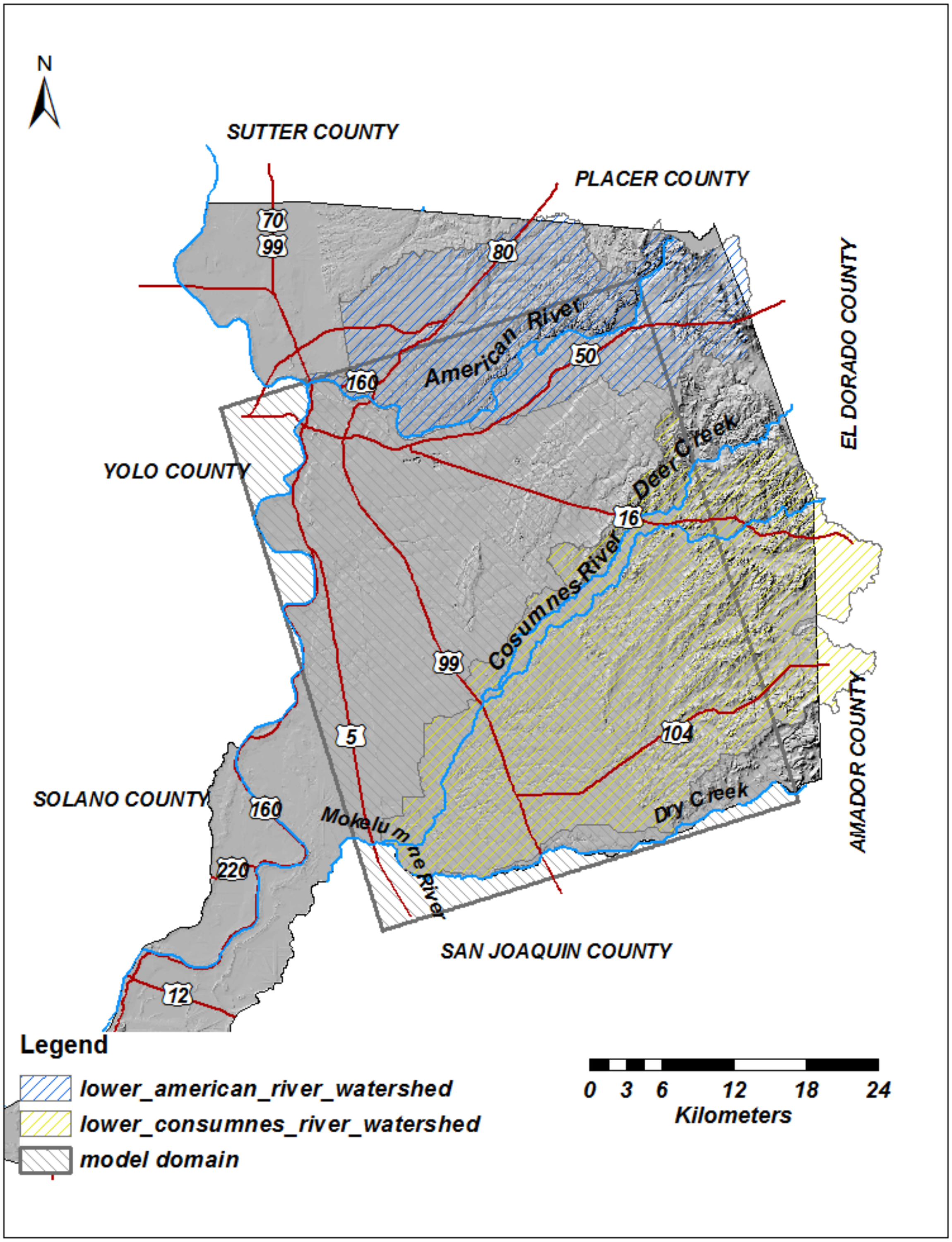


Figure 2.1 Study area major hydrologic features and numeric model domain.

2.2 Study area geology and groundwater

The geology of Sacramento County is characterized by alluvial fan sediments that have been deposited by rivers drain the west side of Sierra Nevada Mountains (Fleckenstein 2004). The major river includes American river, Cosumnes River and Sacramento River (Figure 2.1). The American River drains 4290 square kilometers catchment stretching from Folsom Reservoir to the crest of Sierra Nevada. The alluvial fan deposit of American River has been significantly affected by cyclic Plio-Pleistocene climate change and glaciation in the Sierra Nevada (Meirovitz 2010), thus it contains large portion of coarse grained and high permeable sediments such as gravel and coarse sand. Comparatively, the Cosumnes River drains a much smaller catchment with an area of 1900 square kilometers. The alluvial fan deposit of Cosumnes River experienced little glacial input, resulting in less portion of coarse grained but more fine grained sediments comparing to American River alluvial fan (Meirovitz 2010). The lower Sacramento River bounds the west boundary of Sacramento County. The floodplain deposits carried by the Sacramento River are primarily fine grained sediments.

The groundwater acted as an important source for agricultural, municipal and industrial water uses in Sacramento County historically. For example, form 1962 to 1969, groundwater supply counted for 45 percent of agricultural water demand and 48 percent of municipal and industrial water demand. Intensive groundwater pumping resulted in massive drop of water table and groundwater storage loss. At 1970, the average level of groundwater elevation dropped to 5 feet (1.5 meters) below mean sea level from 30 feet (9 meters) above mean sea level at 1941. The corresponding storage depletion was about 48 million cubic meters annually around the same period of time (1930 to 1968) (DWR 1974). In the Central Valley, the principle geologic formations that yield groundwater are, from oldest to youngest, the Mehrten, Valley Springs and Ione Formations (DWR 1974). Among them, the major water producer is the Mehrten Formation. These deposits are composed of numerous channels of sand, gravels that come from the erosion of volcanic materials (DWR 1974), but are composed of an even greater percentage of silts and clays (Meirovitz 2010), as is typical of Central Valley sediments. The Ione Formation outcrop can be found in northeastern Sacramento County, and is about 400 feet (120 meters) thick in the study area. It dips to the west at about 5 degrees and persisting as far west as the Sacramento River. The Valley Springs Formation is exposed to land surface on the southeastern boundary of the Sacramento Valley and dipping to the west at a rate of about 2 degrees. The thickness of Valley Spring Formation at outcrop ranges from 75 feet (23 meters) to 200 feet (60 meters). The Mehrten Formation is exposed discontinuously on the east boundary of Sacramento County, dipping to the west at about 2 degrees. At the outcrop, the Mehrten Formation is about 200 feet (60 meters), but it thickens westwards to about 400 to 500 feet (120 to 150 meters) at the central of the valley (DWR 1974).

Two distinct characteristics of the geological structure of Sacramento County were observed:

(1)The alluvial fan strata dips westwards at a general angle of 2 degrees starting from the foothills of Sierra Nevada Mountains (DWR 1974, Meirovitz 2010). The thickness of the strata thickens towards the central of the Valley at a depth about 1500 feet (460 meters) (DWR 1974, Fleckenstein 2006);

(2) In geologic history, the streams meandered back and forth across the surface of the valley, thus forming a network of buried stream channels that are embedded in overbank and floodplain silts and clays that constitute the majority of the system. At both shallow (Figure 2.2) (DWR 1974; Shlemon 1967) and deep (Meirovitz 2010) intervals, the ancestral American River channel deposits angled further southwest than the present day course of the river, reaching the present day Cosumnes River and depositing relatively coarse sands and gravels at depth underneath the Cosumnes River deposits (Meirovitz 2010). Those older buried stream channels provide path ways through which the major portion of groundwater moves (Meirovitz, 2010). Pumping from the American River channel deposit at the north may have significant influence on the groundwater quantity at the south Cosumnes River basin. This connectivity also creates potentials of the migration of contaminant from Sacramento Area to Cosumnes River groundwater (Meirovitz 2010). A representative geologic model should capture these important features.

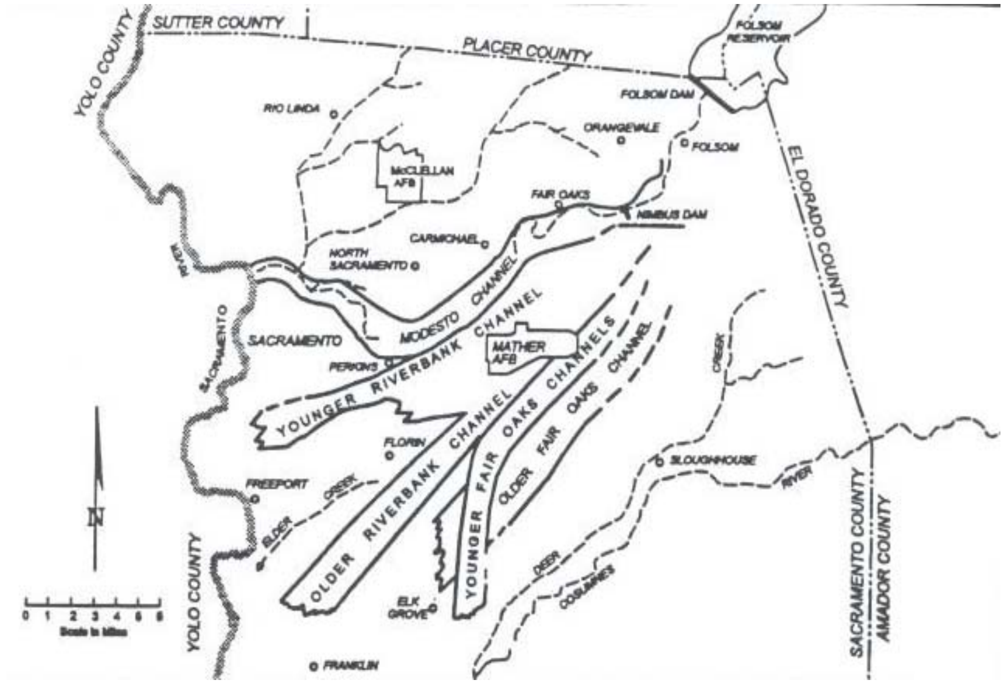


Figure 2.2 The American River older channel locations (Shlemon 1967).

Chapter 3 Methods

3.1 Heterogeneous representation of subsurface

The complexity of the alluvial fan depositional system makes detailed characterization of their heterogeneity difficult (Weissmann and Fogg 1999). Yet, such detailed heterogeneous structures are essentially important for groundwater flow and solute transport model. In general, there are three different methods in characterizing subsurface heterogeneity: the descriptive approach, the processing imitation and the structure imitation (Koltermann and Gorelick 1996). Among the structure imitation method, unlike the traditional empirical curve fitting approach, Carle and Fogg (1996, 1997) and Carle (1997) developed a novel method called the transition probability Markov Chain geo-statistical approach. This new indicator based geo-statistical method has the ability of incorporating subjective geological interpretations such as hydro-facies volumetric proportion, mean length of hydro-facies, juxtapositioning patterns and asymmetry of hydro-facies or even conceptual information, into transition probability Markov Chain model of spatial variability for constructing representative subsurface heterogeneous sediment system, while obeying basic probability rules at the same time (Carle and Fogg 1996, 1997, Carle 1997). Further, by incorporating the transition probability geo-statistical approach within a sequence stratigraphic framework could overcome the tenuous assumption of stationarity that often made when applying geo-statistical approach, and represent multi-scale heterogeneity (Weissmann and Fogg 1999). Weissmann and Fogg (1999) applied this approach on characterizing Kings River fluvial fan system that accounts for multi-scale heterogeneity represented by spatial variable hydro-facies within sequences and spatial variability attribute unique to each sequences and achieved great success. The dominant effect of the subsurface heterogeneity in a system like the Cosumnes River alluvial aquifer complex is that it controls the connectivity of aquifer and aquitard materials, which in turn are controlled by the horizontal/vertical mean length, volumetric proportion of the geologic units. Thus, it would be beneficial to include these geological information along with other subjective geological interpretations, such as observed depositional trend into geological model for a plausible representing of subsurface flow and transport modeling. The conditional simulation modeling software known as TPROGS (Carle and Fogg 1996, 1997, Carle 1997) has the capability of producing such a model, thus is used in this study.

In the study area, Sacramento County, the hydro-facies soil map, lithological logs (Figure 3.1) and driller's logs are the main sources of conditioning data for the geological model development (Meirovitz 2010). The subsurface sediments were categorized into four types, namely gravel, sand, muddy sand and mud, based on its texture (Fleckenstein 2004, Meirovitz 2010). The hydrostratigraphy of the area consists of the interaction of two overlapping alluvial fans, namely American River fan to the north which has a larger volumetric proportion of coarse grained deposits and the comparatively fine grained deposit dominated Cosumnes River fan to the south. Overall, this is a fine grained deposit dominated multi-aquifer system with complex aquifer connectivity. The mean length and the volumetric fraction of hydro-facies of the two fan systems are quite different. Therefore, the study area was divided into north part (American River fan) and south part (Cosumnes River fan) for modeling of the subsurface heterogeneity separately to honor the geological information observed then combined together (Meirovitz 2010). For detailed procedure of the data processing and model development, one can refer to Meirovitz (2010). Ten realizations of the geologic model were generated and one is chosen for this study (Figure 3.2). Table 3.1 lists the volumetric fraction of each hydro-facies in observed data and TPROGS realization (Meirovitz 2010).

Table 3.1 Comparison of volumetric proportion of hydro-facies for data and model.

Comparison of Volumetric Fractions				
	Gravel	Sand	Muddy Sand	Mud
Final Model	0.23	0.14	0.18	0.45
Observed Data	0.20	0.12	0.27	0.41

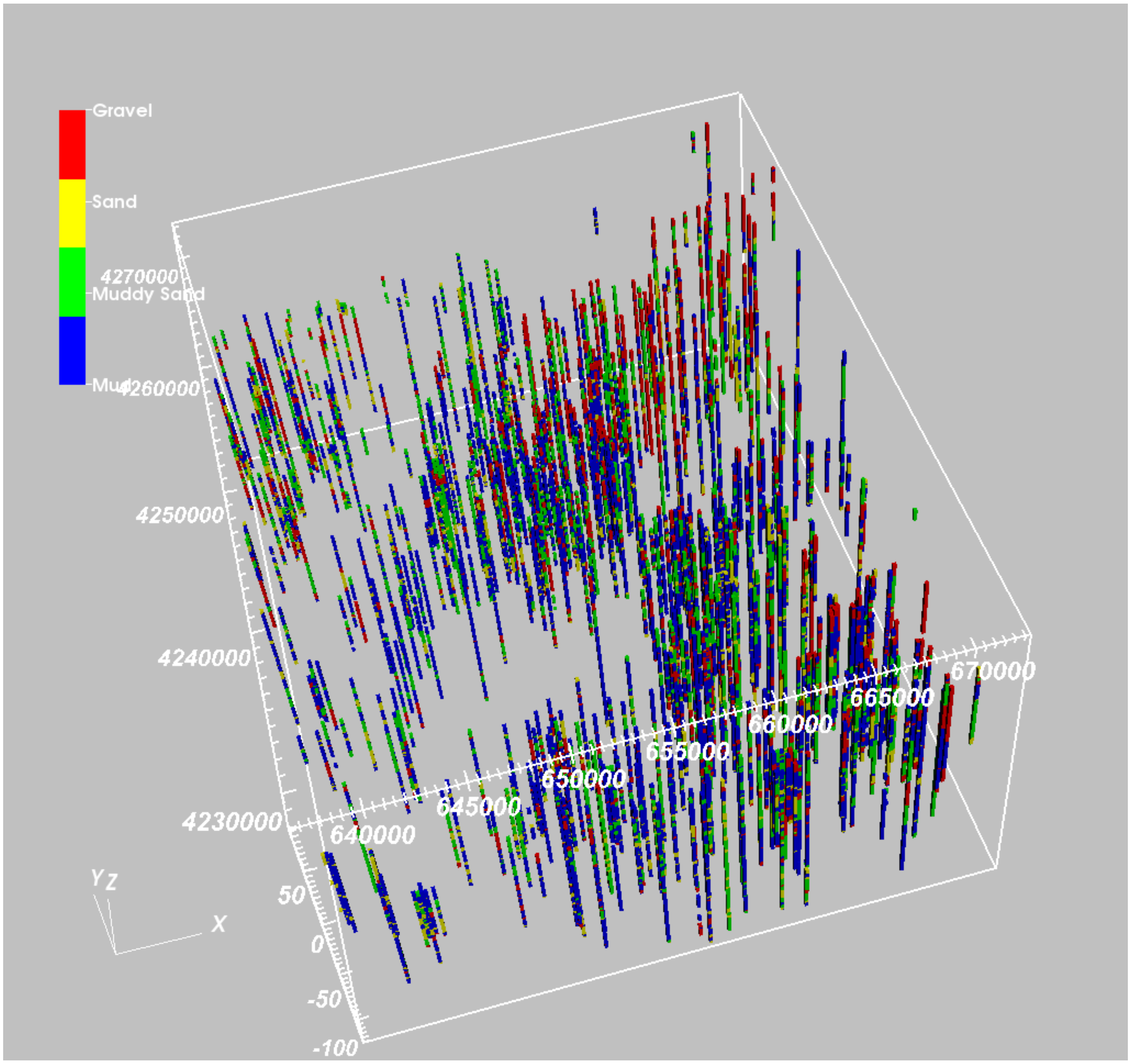


Figure 3.1 Well locations and logs used to develop the geological model (origin (636631.0m, 4228115.5m, -125.0m in WGS84-UTM Zone 10 N coordinate system).

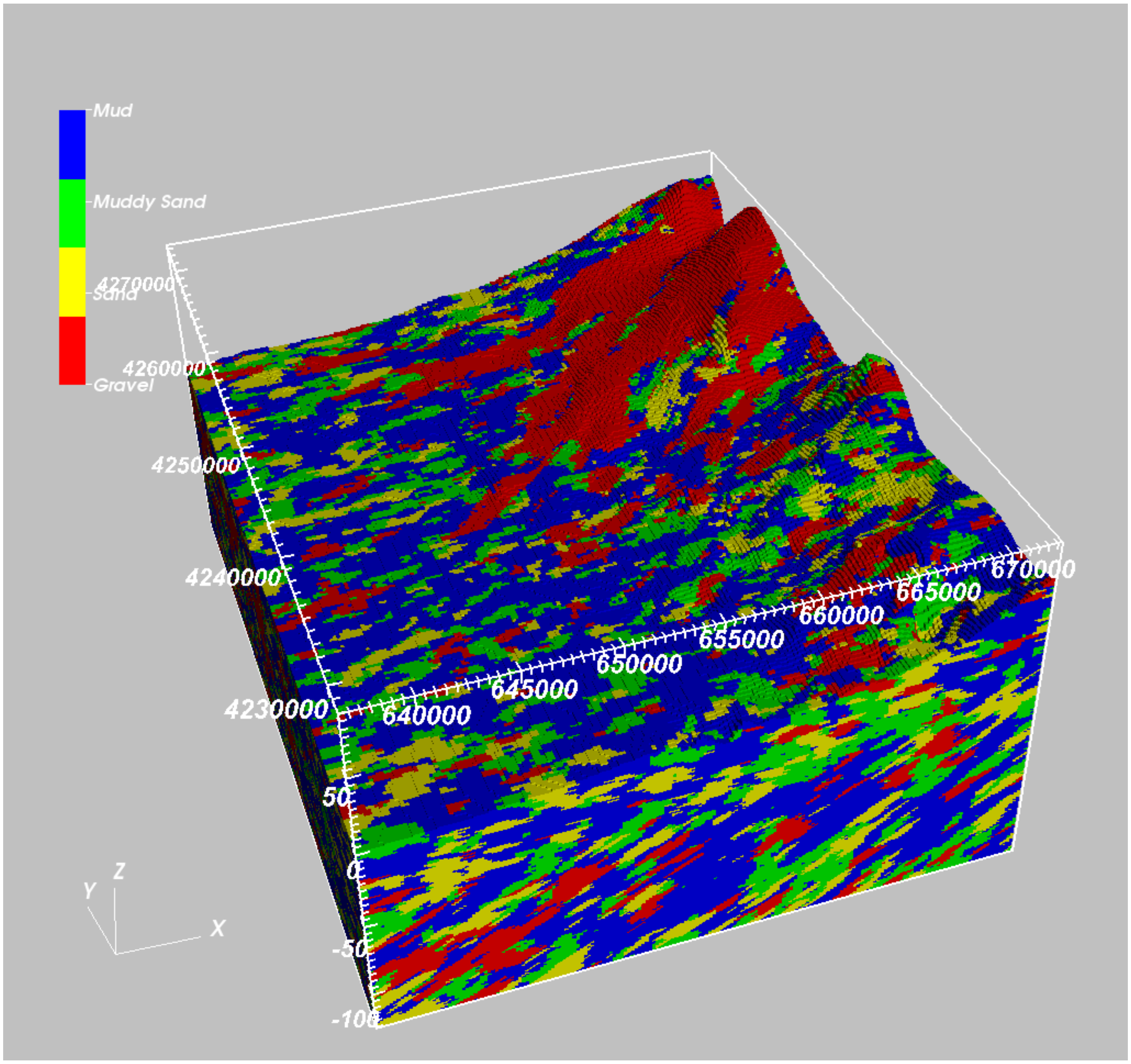


Figure 3.2 One TPROGS realization of subsurface heterogeneity (positive X-axis pointing East; positive Y-axis pointing North; positive Z-axis pointing upward, coordinates are in WGS84-UTM10N coordinate system).

3.2 Integrated hydrologic modeling computer code

Integrated hydrologic modeling of groundwater and surface water interaction in large scale heterogeneous system remains a challenging because of the highly non-linear aspect of partially saturated flow, three-dimensional heterogeneity requiring large systems of equations, and the large contrast in permeability between aquifer and aquitard in geologic systems (Frei 2008, Frei *et al.* 2009). In this study, ParFlow (Ashby and Falgout 1996, Jones and Woodward 2001, Maxwell and Miller 2005, Kollet and Maxwell 2006) is chosen as the computational platform, for it is one of the few available codes which implements a parallel computing scheme and solves Richards Equation for both fully saturated and variably saturated flow in three dimensions with a fully mass conservative manner.

3.2.1 Variably saturated groundwater flow

In ParFlow, the variably saturated flow is represented by Richards Equation, given as equation (1):

$$S(p)Ss\frac{\partial p}{\partial t}+\frac{\partial(S(p)\rho(p)\emptyset)}{\partial t}+\nabla\cdot\left(K(p)\rho(p)(\nabla p-\rho(p)g\nabla z)\right)=Q,\text{in }\Omega \qquad (1)$$

Where Ω is the flow domain; p is the pressure head [L]; S is the soil water saturation; Ss is the specific storage of porous media [1/L]; ϕ is the porosity of the porous media; $\boldsymbol{K(p)}$ is the soil saturation dependent hydraulic conductivity tensor [L/T]; $\boldsymbol{g}$ is gravity vector [L/T^2];Q is resource or sink term [L^3/T].

The hydraulic conductivity tensor is calculated as equation (2):

$$K(p)=\frac{KKr(p)}{\mu} \qquad (2)$$

Where K is the saturated hydraulic conductivity of porous media; and $Kr(p)$ is the relative hydraulic conductivity coefficient; μ is the water viscosity (ParFlow Manual 2010).

The soil retention and relative hydraulic conductivity functions are represented by the Van Genuchten formula as equation (3) and (4):

$$S(p)=\frac{Ssat-Sres}{(1+(\alpha p)^{n})^{(1-\frac{1}{n})}}+Sres \qquad (3)$$

$$Kr(p)=\frac{(1-\frac{(\alpha p)^{n-1}}{(1+(\alpha p)^{n})^{1-\frac{1}{n}}})^{2}}{(1+(\alpha p)^{n})^{\frac{1-1/n}{2}}} \qquad (4)$$

Where *Ssat* [-] is the relative saturated water content; *Sres* [-] is the relative residual water content; α [1/L] and n [-] are empirical fitting parameters (Van Genuchten 1980).

3.2.2 Overland flow

ParFlow can simulate fully coupled surface flow and subsurface flow via an overland flow boundary condition. The shallow overland flow is represented by the continuity equation and the momentum equation.

In two dimensions, the general form of the continuity equation for surface water flow is as equation (5):

$$\frac{\partial \varphi_s}{\partial t} = \nabla \cdot (v\varphi_s) + q_r(x) \qquad (5)$$

Where φ_s is the surface water ponding depth [L]; t is the time [T]; v is the depth averaged velocity vector [L/T]; q_r is the general source/sink term [L/T].

The general form of momentum equation is given as equation (6):

$$\frac{\partial v}{\partial t} + v\frac{\partial v}{\partial x} + g\frac{\partial \varphi}{\partial x} = g(S_f - S_o) \qquad (6)$$

where v is depth averaged velocity [L/T]; g is gravity; S_f is the friction slope [-]; S_o is the bed slope [-]. When the diffusion term is ignored (Kinematic Wave Approximation), the formula can be written as $S_f = S_o$.

The depth discharge relation is given by empirical Manning's formula as equation (7):

$$V = \frac{\sqrt{S_f}}{n}\varphi^{2/3} \qquad (7)$$

Where n is the Manning's roughness coefficient [T/L$^{1/3}$]; V is depth averaged velocity [L/T].

ParFlow implements a form of the continuity equation that could be directly coupled to the system of equation via the boundary condition at the land surface. This form of continuity equation in ParFlow is as equation (8):

$$\frac{\partial \varphi_s}{\partial t} = \nabla \cdot (v\varphi_s) + q_r(x) + q_e(x) \qquad (8)$$

Where the term q_e is the exchange flux term between surface water and groundwater [L/T]; the rest terms are defined the same as above (Kollet and Maxwell 2005).

Many of other research on the surface water and groundwater flux exchange is based on the conductance concept, that is equation (9):

$$q_e(x) = \lambda(x)(\varphi_s - \varphi_p) \qquad (9)$$

Where φ_p is the subsurface pressure head [L]; φ_s is the surface pressure head*;* $\boldsymbol{\lambda}$ is conductance coefficient. This approach assumes there exists a distinct interface between surface and subsurface, where the conductance coefficient $\boldsymbol{\lambda}$ comes into play. However, it is very difficult to determine the value of $\boldsymbol{\lambda}$ from direct field observation.

Instead, ParFlow calculates the exchanged flux by including the overland flow equation into the Richards Equation at the top boundary cell under saturated conditions. Using condition of continuity of pressure $\varphi_s = \varphi_p = \varphi$ and flux $q_{bc} = q_e$ at the ground surface, the exchang flux term could be calculated as equation (10):

$$q_e(x) = \frac{\partial \| \varphi, 0 \|}{\partial t} - \nabla v \parallel \varphi, 0 \parallel - q_r(x) \qquad (10)$$

This flux is substituted into the Neumann type boundary condition, stated as equation (11):

$$-KK_r(p)\nabla(p - z) = q_{bc} \qquad (11)$$

Then equation (11) combines with equation (10) results in equation (12):

$$-KK_r(p)\nabla(p - z) = \frac{\partial \| \varphi, 0 \|}{\partial t} - \nabla v \parallel \varphi, 0 \parallel - q_r(x) \qquad (12)$$

Where $\parallel A, B \parallel$ indicates the greater value between A and B.

This formula results in the surface water flow equation functioning as a boundary condition for the Richards Equation (Kollet and Maxwell 2005).

3.3 Model set up

3.3.1 Model discretization

The model domain covers a 36200 meters (east-west) by 45400 meters (north-south) region that includes Cosumnes River alluvial fan system and portion of American River alluvial fan system. The heterogeneous geological model generated by TPROGS is 125 meters deep due to the limited available data (Meirovitz 2010). The aquifer systems in the study area deepen westwards from the Sierra Nevada Mountains foothills, where the aquifers are 500 feet (150 meters) deep. At the center of the valley, along Sacramento River, the aquifer can be as deep as 2000 feet (600 meters) (Fleckenstein 2004, RMC 2011). Therefore, Additional 50 homogeneous layers were added to the bottom of the heterogeneous geological model for properly incorporating deeper aquifer system. Cell sizes of the model are 200 meters in x and y directions and 1 meters in the vertical direction. This result in the total number of computational cells of the model is slightly over 10 million. Digital elevation model data was mapped onto the surface of the domain (Figure 3.7).

3.3.2 Boundary conditions and initial conditions

Correct selecting of boundary conditions is a critical step in groundwater flow model design, and this step is most subject to serious errors. Considering the purpose of this study, the hydrologic conditions of the study area and the availability of data, boundary conditions for the model is carefully selected and justified (Figure 3.7).

The east boundary of the model domain lies close to the foothill of Sierra Nevada Mountains, where the groundwater level varies moderately year to year. Therefore, specified head boundary condition is applied at the east boundary. Fall groundwater elevation data from California Department of Water Resources (DWR) monitoring wells for years 1977, 1984 and 2004 were kriged (Figure 3.4, 3.5 and 3.6). The groundwater elevation at the east boundary for the three selected years are averaged and assigned as specified head boundary condition (Figure 3.3). Year 1977, 1984 and 2004 was chosen, for they represent dry, wet and average hydrologic conditions of the study area (RMC 2011).

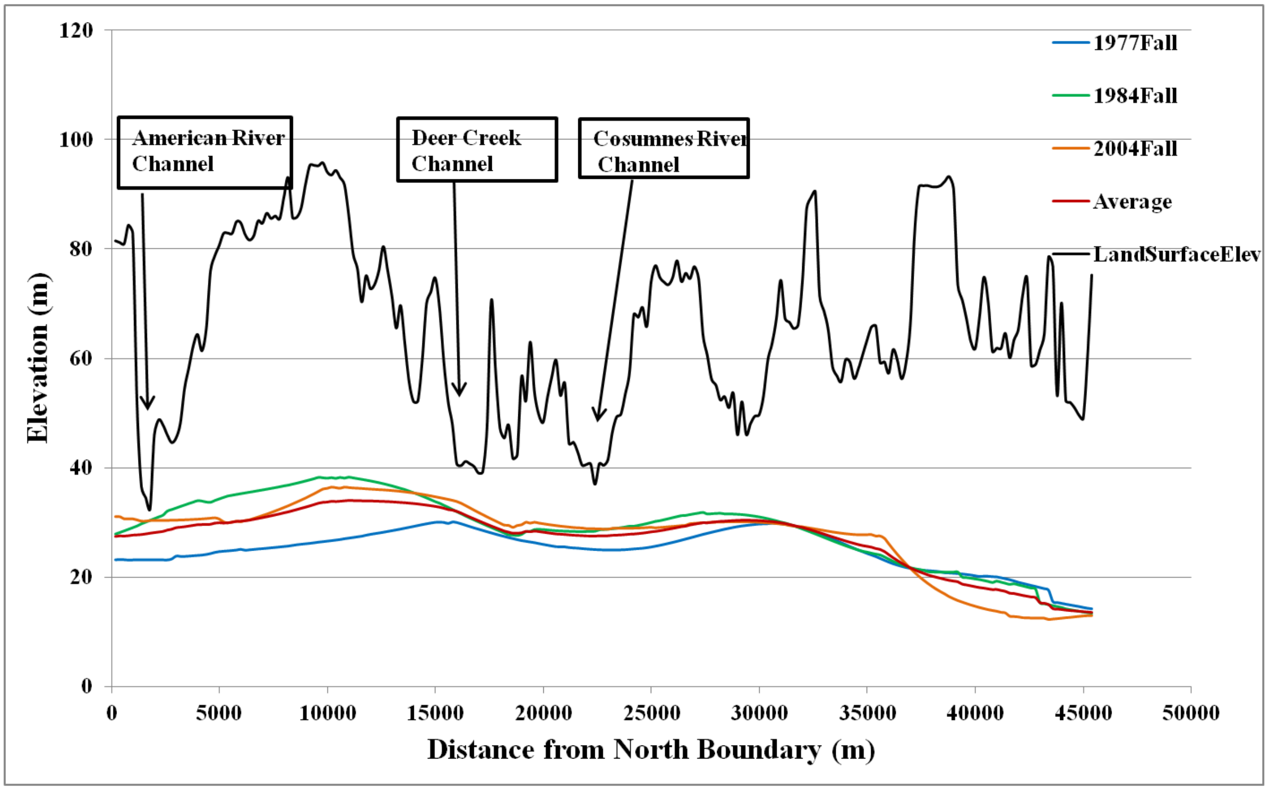


Figure 3.3 Specified head boundary condition for east boundary (note: the land surface elevation was taken from the re-sampled 200 meters DEM, which is too coarse to represent land surface details).

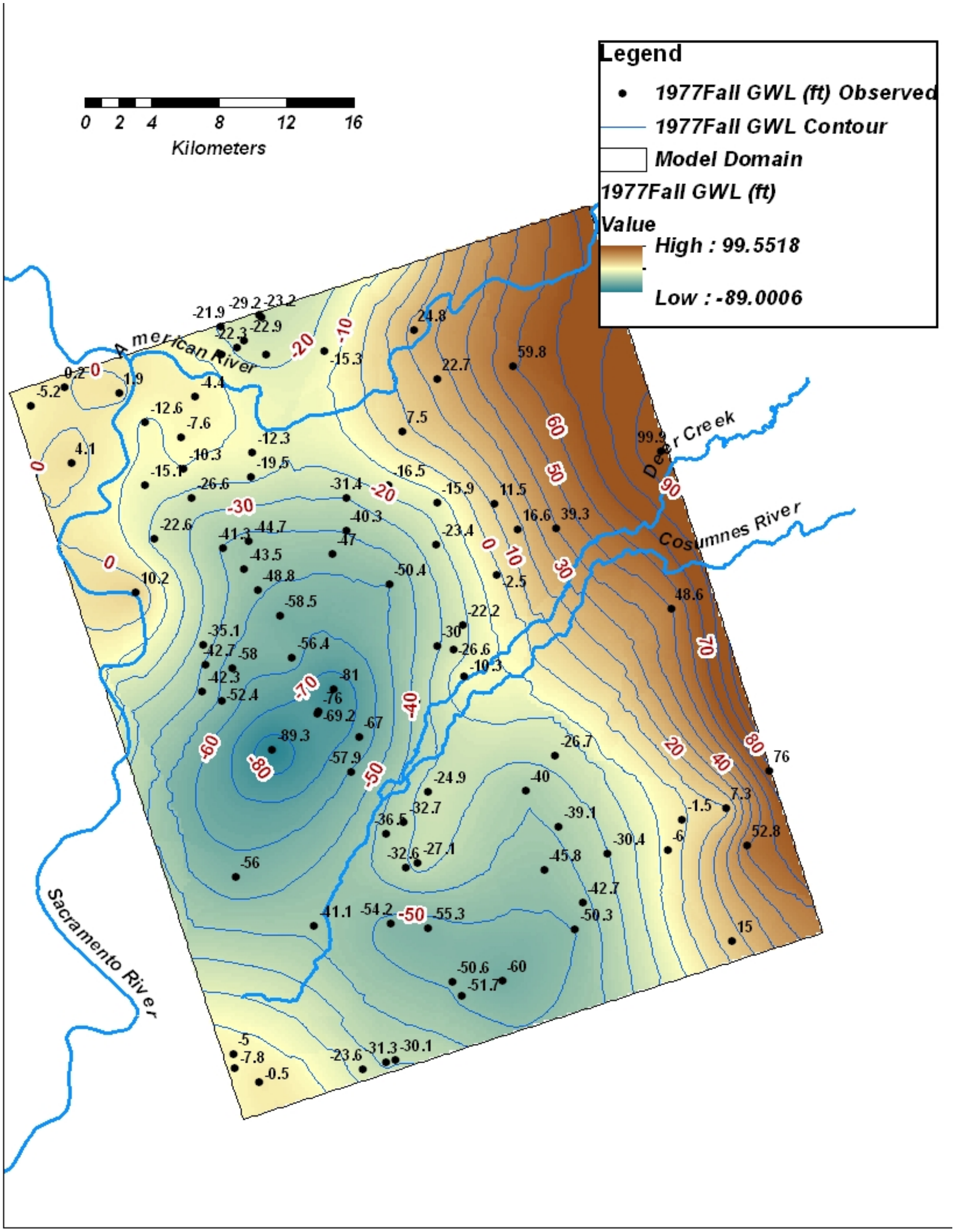


Figure 3.4 1977 fall groundwater levels.

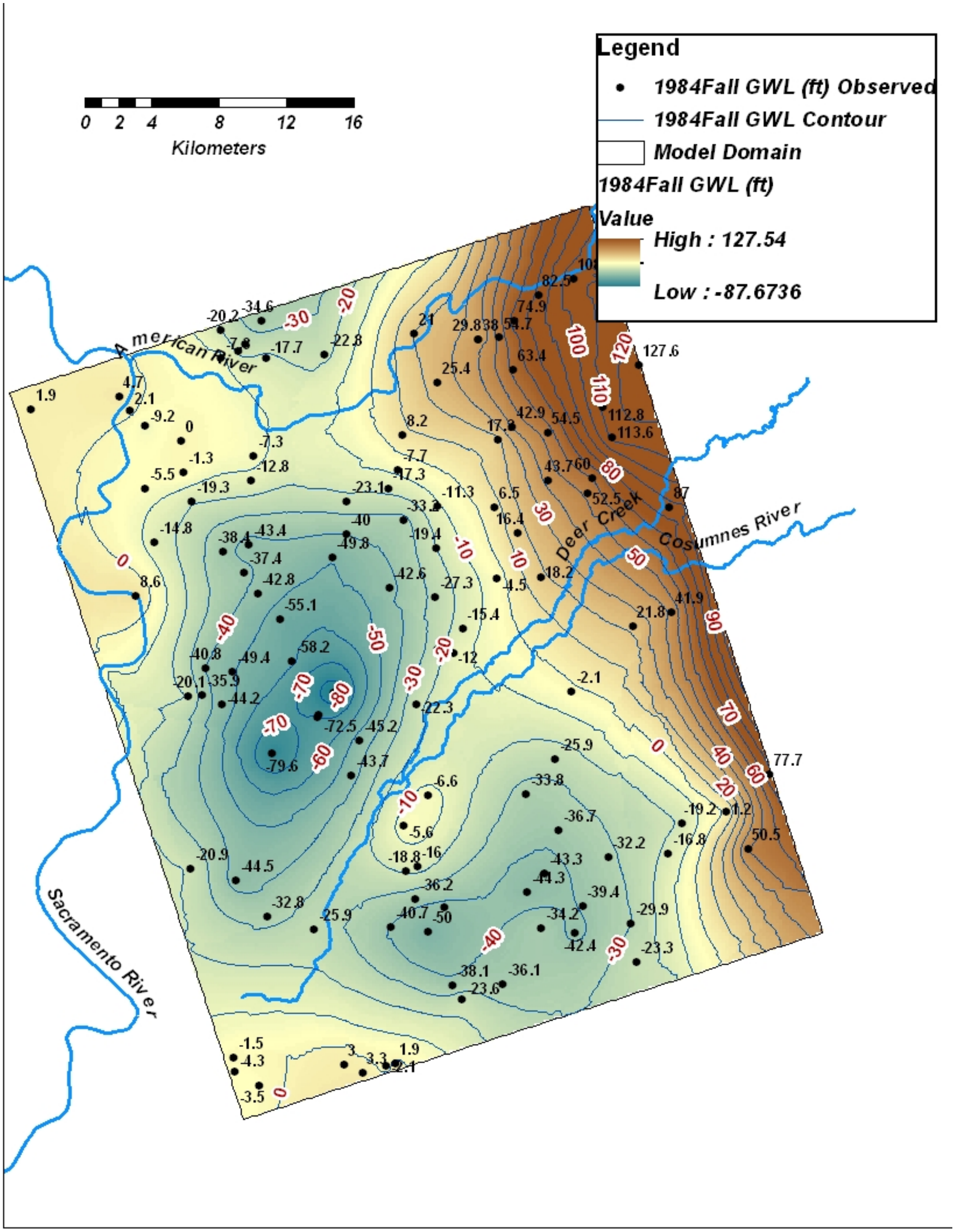


Figure 3.5 1984 fall groundwater levels.

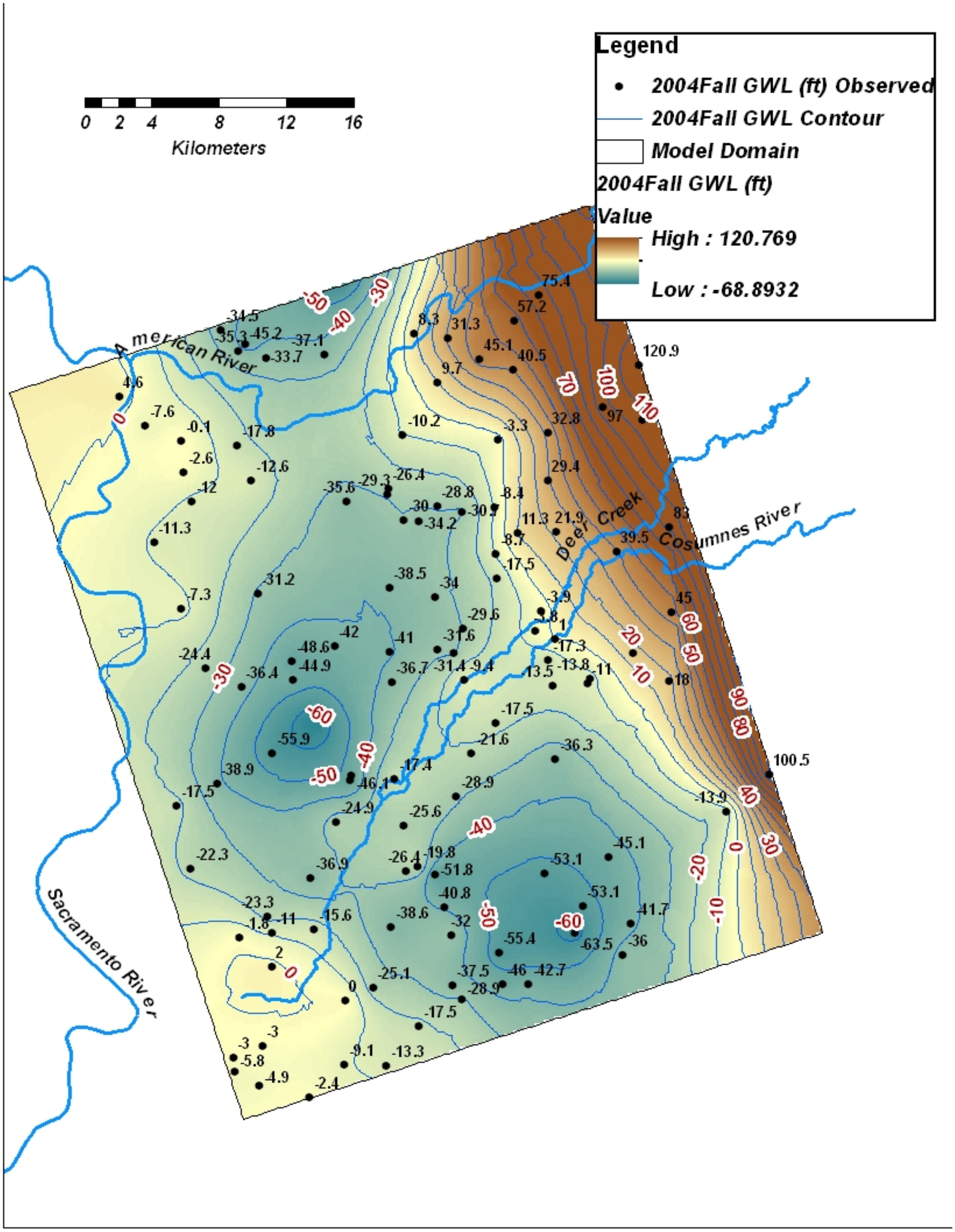


Figure 3.6 2004 fall groundwater levels.

The west boundary of the model domain is along the Sacramento River approximately. A distal-head boundary condition (Kipp 1986) is applied to represent the effect of the Sacramento River and the Sacramento-San Joaquin Delta on groundwater conditions on the west boundary. The southwestern part of the study area is close to Sacramento-San Joaquin Delta, where the water level is around mean sea level with mean value changing by 1 foot annually and 2-3 feet inter annually (Personal Communication, 2013, William Fleenor). Therefore, the general head value is set to be mean sea level at 1000 meters west of Sacramento River. This allows groundwater head to vary along the western boundary, but with the general constraint of stable, sea-level heads west of that boundary.

The general form of head dependent boundary condition is given as equation (13)

$$Q = KA\frac{h_{source} - h_{model}}{b} \qquad (13)$$

Where Q is the volumetric flux [L^3/T]; A is the cross section area of water flows [L^2]; h_{source} is the head specified on the boundary [L]; h_{model} is the head calculated by the model on the boundary [L]; b is the distance between where h_{source} is specified and the model boundary [L]; K is the hydraulic conductivity of the material between where h_{source} is specified at the model boundary [L/T]. The term $\frac{h_{source} - h_{model}}{b}$ is called conductance term (Anderson and Woessner 1992). In model packages, such as MODFLOW, the general head boundary condition is implemented by changing the conductance term. Usually the change in conductance term is achieved by changing the distance b. However, in current version of ParFlow, variable cell discretization on horizontal direction is not implemented. To implement general head boundary condition in ParFlow, the hydraulic conductivity of the cells on west boundary is adjusted to reflect the conductance change. The adjusted hydraulic conductivity is calculated as equation (14),

$$K_{new} = K_{old}/(\frac{1000}{ddx}) \qquad (14)$$

Where ddx is the cell width on horizontal direction, K_{new} is the adjusted hydraulic conductivity, K_{old} is the original hydraulic conductivity.

The top boundary is specified flux boundary condition with the flux to be deep percolation estimated from the regional integrated water resource management (SACIWRM) model (RMC 2011). Specified head boundary condition is applied on the streams, including Cosumnes River, American River and Deer Creek. The stream stages are estimated from SACIWRM model stream nodes and vary weekly. More details on data processing, one can refer to appendix A.

With added layers to represent deep aquifers, the model includes major production aquifers in Sacramento County. Beneath the deep aquifers are much lower hydraulic conductivity rocks (RMC 2011), thus no flow boundary condition is applied to the bottom. The north and south boundary locate close to streams (e.g. American River to the north, Dry Creek to the south), thus, flux across the boundaries is expected. However, the main focus area of this study is on Cosumnes River basin, which locates in the middle of the model, and far away from both north and south boundaries. Therefore, the flux across north and south boundaries will not have profound effect of the hydrologic and hydraulic response of Cosumnes River basin. To simplify the model setup, no flux boundary condition is applied to both north and south boundaries. In the future, head depended boundary condition should be applied to north and south boundaries to better represent field condition appropriately.

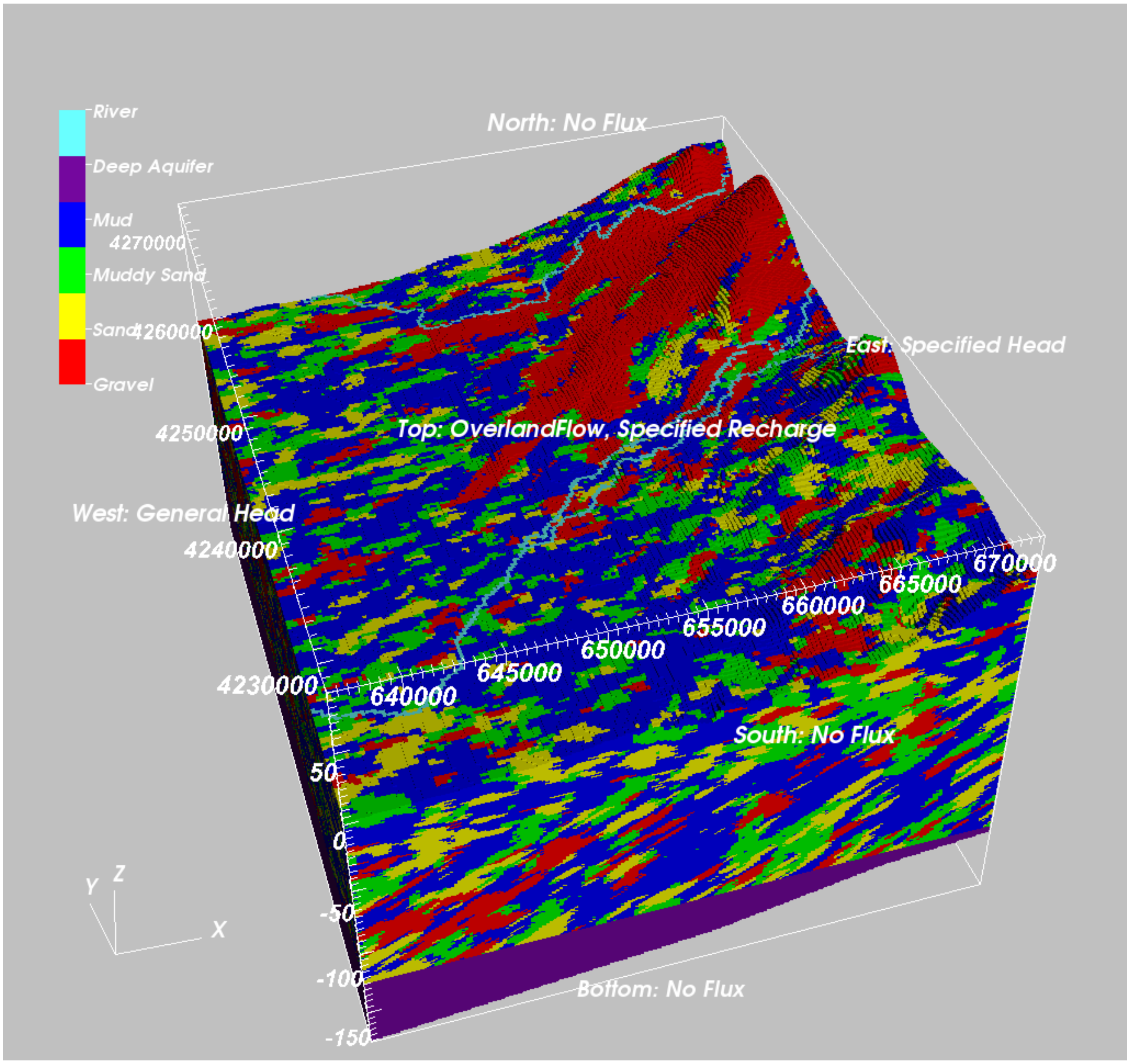


Figure 3.7 Numerical model domain and boundary conditions (the bottom purple segment represents added layers for representing deeper aquifer system. Coordinates are in WGS84-UTM10N coordinate system).

1969 fall (September and October) groundwater head from DWR monitor wells was kriged to obtain the initial distribution of groundwater head for calibration runs (Figure 3.8).

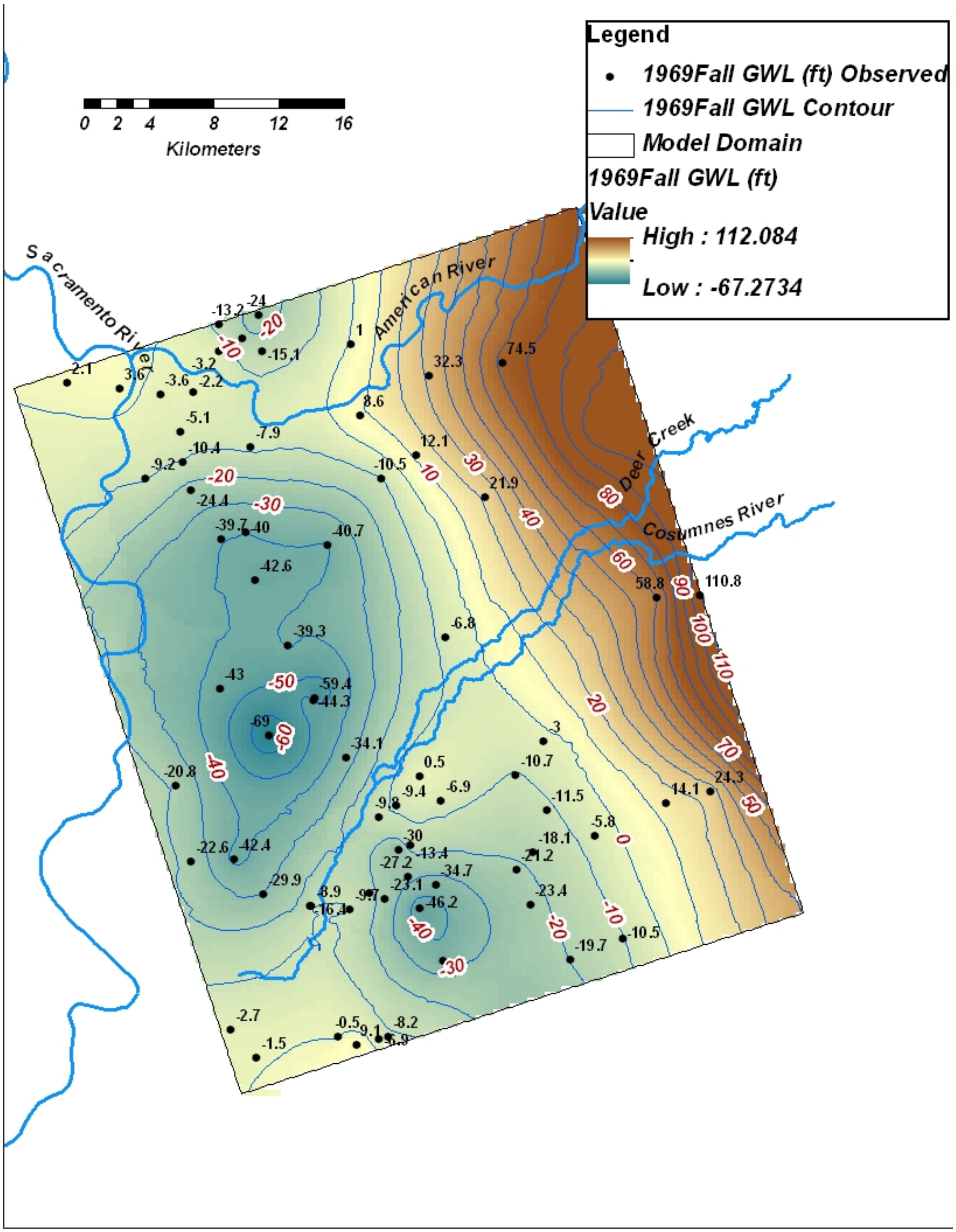


Figure 3.8 1969 fall initial groundwater head condition.

3.4 Model calibration

Water year 1969-1985 was chosen as the calibration period, for this period includes dry year (1977), wet year (1984) and normal years. The calibration was performed in transient mode, because a regional scale groundwater system which is under intense groundwater pumping, agricultural practice, and human activities is always transient. Inverse technique (UCODE) was initially used to estimate the hydraulic parameters of each hydro-facies. However, it results in un-acceptable computing time due to the complexity and transient characteristics of the model as well as limited accessible computational resources. Therefore the hydraulic parameters of each hydro-facies were adjusted manually in the calibration process. Among many of the hydraulic parameters of hydro-facies, the hydraulic conductivity is found to be the most sensitive parameter that affects the groundwater flow system (Sager 2012). Other parameters such as Van Genuchten parameter, porosity and specific storage are relatively less sensitive (Sager 2012). Thus hydraulic conductivity of hydro-facies is the focusing variable in the calibration process, while other parameters stay unchanged. The initial estimate of all the parameters of hydro-facies is summarized in Table 3.2. One can refer to appendix C on details about the estimation of deeper aquifer hydraulic parameters.

In this study, the groundwater budget is adopted from the regional integrated water resource management model SACIWRM (RMC 2011). In the calibration process, the groundwater pumping for urban regions (Figure A.3) is kept unchanged, while the groundwater pumping for agricultural regions (Figure A.3) is adjusted when necessary in order to achieve the calibration goal. These adjustments in pumping rates for agricultural region are justified in this study because the pumping and recharge estimates from the SACIWRM model stem from a calibration of that model, which is based on a conventional, essentially homogeneous model. Because the current model has a very different, but more realistic geologic structure, and because the crop consumptive use method of estimating groundwater recharge (used in SACIRWM) is only approximate, the adjustments of pumping of agricultural region in this study is OK, as long as the adjustment falls within the estimated error bounds of crop consumptive use method (typically less than 20-50 percent). The calibration is to determine whether the heterogeneous model is plausible not only in the context of available geological data, but also the water budget information as presented by SACIWRM. In other words, the purpose of the calibration in this study is not to develop a predictive model, but rather to determine whether the heterogeneous model is both plausible and consistent with the water budget, accounting for the fact that the water budget itself is approximated.

The calibration process is a two-step procedure. The first calibration goal is to examine whether the vertical connectivity, or vertical effective hydraulic conductivity of the model is consistent with field conditions. In multi-aquifer systems like the Cosumnes River alluvial fan system, the shallowest portions behave like unconfined or semi-confined aquifers, with relatively moderate seasonal changes in head owing to the influence of unconfined groundwater storage mechanisms that are dominated by the specific yield (S_y) parameter, resulting in smaller changes in head for a given change in groundwater storage. In contrast, the deeper aquifer zones are increasingly confined by multiple mud and muddy sand confining layers, resulting in much more dramatic seasonal changes in head that are controlled by the much smaller (as compared to S_y) specific storage (S_s) coefficient. Herein, the shallow zone will be referred to as "semi-confined," and the deeper zone as "confined." In reality, the entire system is characterized by semi-confined conditions that become increasingly confined with depth. If the vertical connectivity of the aquifer materials is too great, or the hydraulic conductivity of the confining beds is too high, the seasonal fluctuations in groundwater levels in the deeper, more confined portions will be too small, and will look too muted and similar to that of the shallower sections. Accordingly, the first step in calibration is to examine whether shallow to deep trend of semi-confined to confined-type well hydrographs is preserved. Note that there are no long-term records of the shallow semi-confined fluctuations because virtually all of the DWR monitoring wells are deep enough to be in the confined section. However, experience in other parts of the Central Valley shows that the semi-confined groundwater head fluctuates seasonally only on the order of meters, compared to 10's of meters for the deeper confined sections, depending on the pumping rate.

The second calibration goal is to examine whether the simulated groundwater head seasonal variation show good matches with the observed seasonal groundwater head variation at selected calibration wells.

Table 3.2 The initial estimates of parameter values.

Hydro-Facies	Hydraulic Conductivity	Specific Storage	Porosity	VG-Alpha	VG-N	VG-Residual Saturation	VG-Saturated Saturation
Gravel	45.26 m/day	2.0e-5	0.35	3.55	3.16	0.1	1.0
Sand	27.12m/day	5.0e-5	0.35	3.55	3.16	0.1	1.0
Muddy Sand	0.1m/day	1.0e-4	0.4	2.69	2.0	0.1	1.0
Mud	0.001m/day	1.0e-3	0.45	1.62	2.0	0.2	1.0
Deep Aquifer	83.17m/day	4.8e-4	0.35	3.55	3.16	0.1	1.0

The boundary conditions of calibration runs are the same as the boundary conditions described in section 3.3.2. The top surface is divided into two patches, namely the stream patch, including American River, Cosumnes River, Deer Creek and the non-stream patch that covers the rest of the land surface. The non-stream patch is applied specified flux boundary condition with the flux to be deep percolation from SACIWRM model (RMC 2011). The stream patch was initially applied overland flow boundary condition (section 3.2.2) with daily variable river discharge from USGS gage station on American River, Cosumnes River and Deer Creek. But, implementing overland flow boundary condition results in very long computing time, due to the frequent changing of river discharge and the complexity of the model. Therefore, the boundary condition for the stream patch is switched to be specified head boundary condition, with the stream stages taken from SACIWRM model stream nodes.

The initial condition of the calibration runs is the potentiometric surface interpolated from 1969 fall (September and October) groundwater head of DWR monitoring wells (Figure 3.8).

3.5 Floodplain inundation simulation scenarios

The groundwater has long been an important component of California water resources. Its role is even more profound during drought. For example, during 1988-1992 drought, there was a 25 percent annual increase in the sales of pumps. The groundwater pumping accounted for a significant portion of water supplies in that period (Zilberman *et al.* 1995). Appropriately managed groundwater basin can act as a water buffer for drought. Its role on buffering drought and secure California water supply can be more significant considering the climate change impacts. Capturing and infiltrating winter storm water into local aquifer or cone of depressions that was created by massive groundwater pumping is a feasible strategy for aquifer recharging and groundwater restoration, but it is not commonly found in practice (Langridge *et al.* 2012). This practice can be very important for developing local groundwater reservoir that can potentially mitigate effects of snow and surface storage decline due to drought and climate change.

Groundwater surface water interaction is the key process for understanding the mass exchange between surface water and groundwater. Understanding groundwater flow mechanism in heterogeneous variably saturated vadose zone is beneficial for proper managing vadose zone and evaluating the winter floodwater recharging practices. Eventually improve conjunctive groundwater surface water management to address the climate change challenges. In this study, groundwater and surface water interaction under floodplain inundation at lower Cosumnes river floodplain is simulated, in hoping the results will provide insights on mechanism of groundwater and surface water interaction in heterogeneous vadoze zone and be helpful on guiding the development of local groundwater reservoir. In addition, in hoping that this study can guide Cosumnes River protecting and restoration practices, as well as manage similar fluvial fan aquifer system.

3.5.1 The geological setting of floodplain

The lower Cosumnes River floodplain in this study is the same as the floodplain in RBI 2013 FLO-2D floodplain inundation modeling (Figure 3.9A). This area primarily locates in The Nature Conservancy Cosumnes Preserve, and spreads about 10 square miles (26 square kilometers) (RBI 2013). But, this is only a small portion of the Cosumnes River basin. The four types of hydro-facies surface area fraction and subsurface volumetric fraction of the floodplain is calculated and summarized in Table 3.3. As shown in Table 3.3, the floodplain is dominated by low permeable muddy sand and mud, which accounts for over 70 percent of the surface area and subsurface volume. Details of the surface heterogeneity and subsurface heterogeneity of the floodplain are shown in Figure 3.9B and Figure 3.9C, respectively.

Table 3.3 Surface area fraction and subsurface volumetric fraction of hydro-facies of floodplain that is depicted in Figure 3.9A.

Hydro-facies	Gravel	Sand	Muddy Sand	Mud
Surface area fraction	7.78%	14.25%	20.38%	57.58%
Subsurface volumetric fraction	10.46%	14.52%	21.83%	53.18%

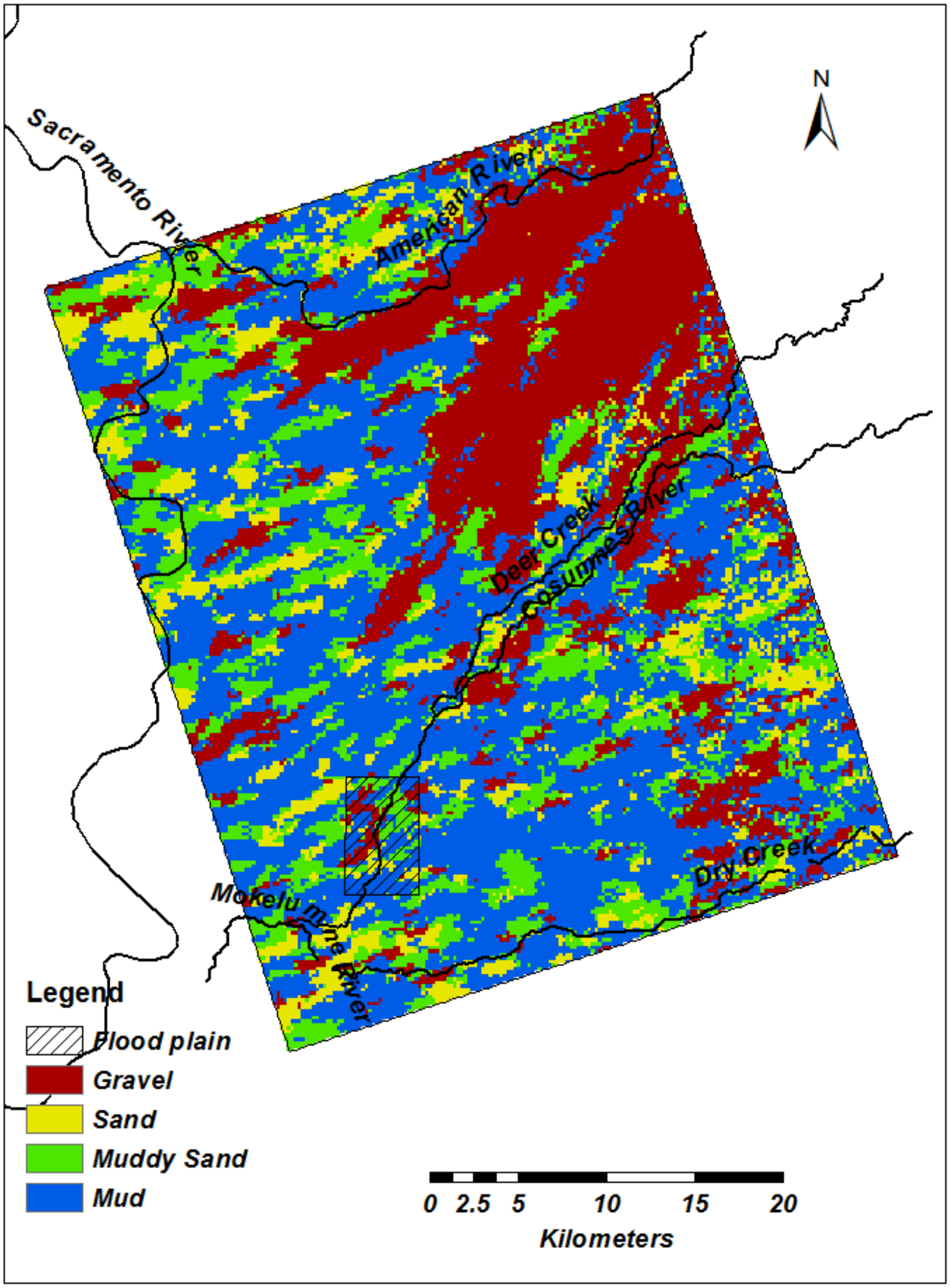


Figure 3.9A Surface heterogeneity of the geological model.

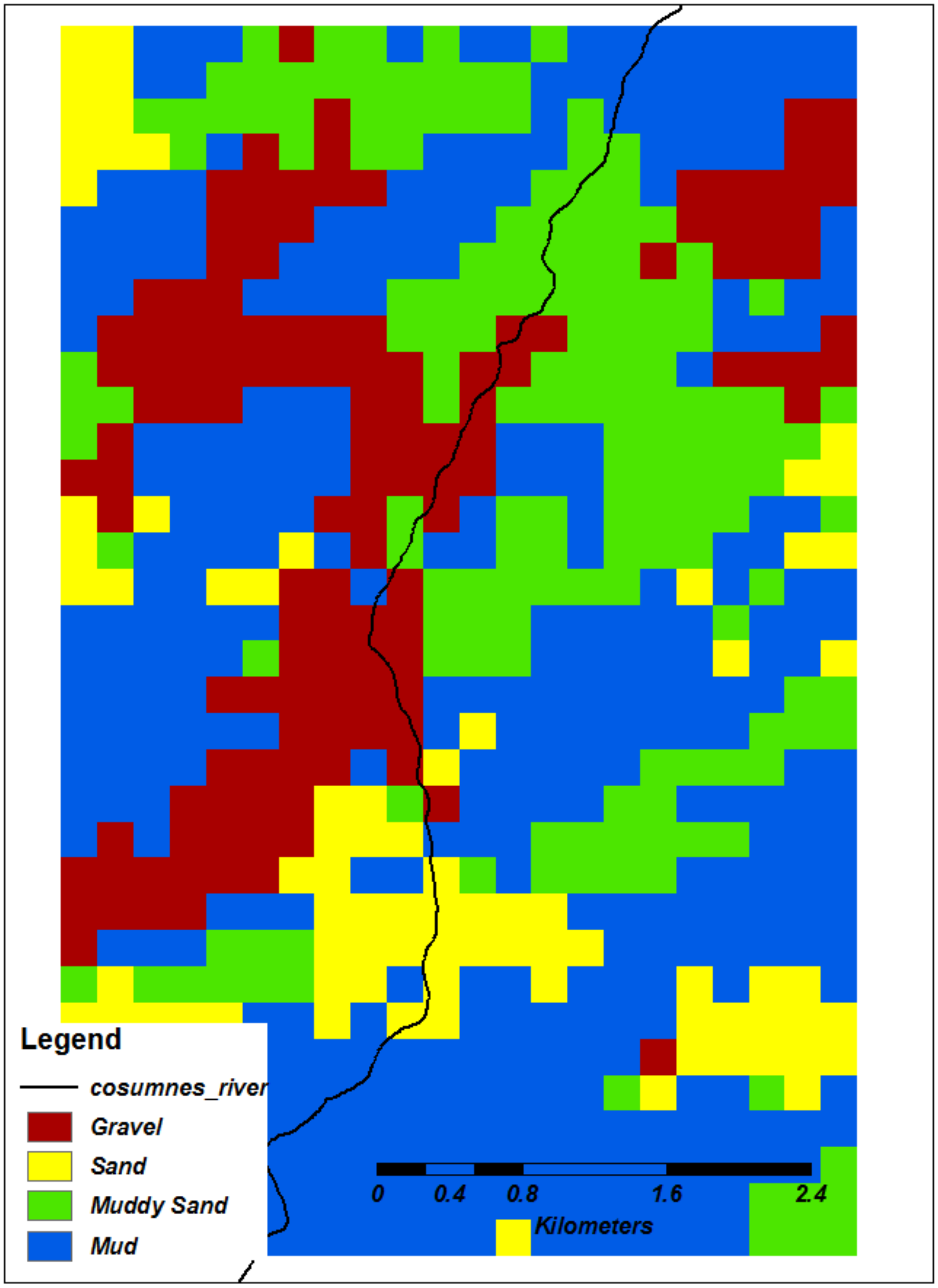


Figure 3.9B Detail of area indicated in A.

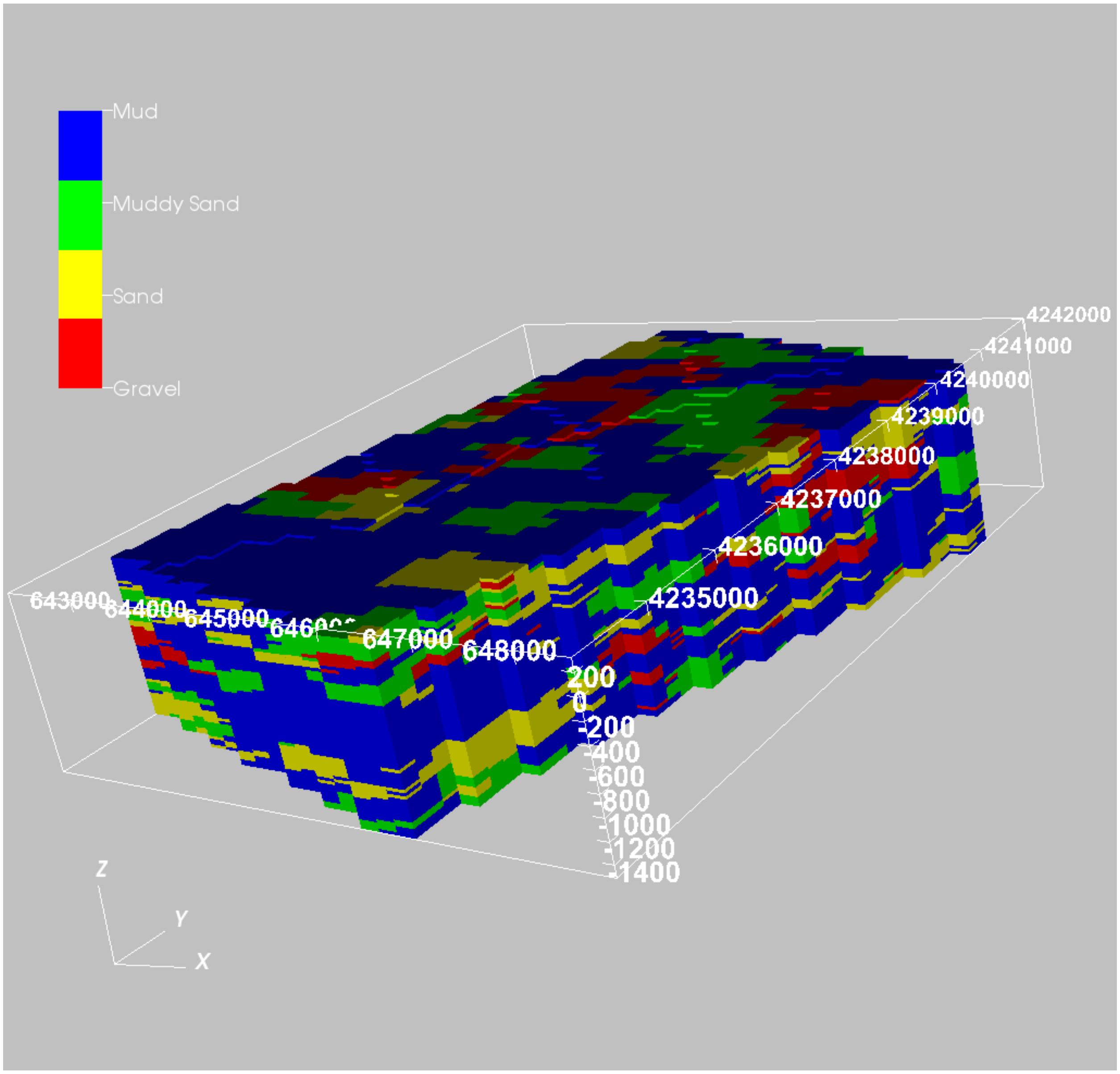


Figure 3.9C Floodplain subsurface geological setting (the coordinates are in WSG84-UTM10N coordinate system, vertical 30x).

3.5.2 Initial and boundary condition

The boundary conditions of the floodplain inundation simulation runs are the same as the boundary conditions described in section 3.3.2. The top surface is divided into two patches, namely the floodplain patch and non-floodplain patch that covers the rest of the land surface. Specified head boundary condition is chosen for the floodplain patch, where inundation depth and spreading is from RBI 2013 floodplain flooding maps. The non-floodplain patch is set to be no flux boundary condition to exclude other sources of recharge from surface.

A spin up model is run for a year to get a proper initial soil moisture distribution. The boundary conditions for the spin up run are the same as described in the previous paragraph except that the floodplain patch is set as overland flow boundary condition with 0.001 meter per day light rain for a year. The initial condition for the spin up model is the same as the one in calibration runs (Figure 3.8).

It worth pointing out that the one year spin up run may not be long enough for the soil moisture to reach steady state, especially in low permeable sediments such as muddy sand and mud. More likely, it may take years to decades for the soil moisture in muddy sand and mud to reach steady state. However, decades spin up run is computational expensive and not conducted in this study, leaving this a potential improvement of the model in the future.

3.5.3 Floodplain inundation scenarios

Groundwater and surface water interaction is modeled under four floodplain inundation scenarios at the lower Cosumnes River floodplain. Each scenario and simulation period is summarized in Table 3.4.

Table 3.4 Floodplain inundation scenarios and time frame.

Scenarios	Peak flood (cfs)	Peak flood time (hour)	Simulation period (hours)
3.5 year flood	12000	25	140 (6 days)
10 year flood	35000	55	432 (18 days)
100 year flood	89000	105	1008 (42 days)
3.5 year flood extend	12000	25	4460 (186 days)

Three types of Cosumnes River flood events from the FLO-2D model are shown in Figures 3.10A, B and C (RBI 2013). The floodplain 3.5-year inundation maps (e.g., inundation depth, spreading time frame) from RBI (2013) are given in Figures 3.11A, B and C. One can refer to appendix A for the floodplain inundation maps of 10 year and 100 year flood events. One can also refer to RBI (2013) final report for more detailed information on flood model construction and simulation. The inundation map is mapped to the ParFlow groundwater and surface water interaction model surface appropriately. It is worth noting that only the flood hydrograph of 3.5 year flood event was long enough (140 hours) to fully simulate the inundation and recession process (RBI 2013). It takes significantly longer time for the 10 year flood event and 100 year flood events to recede, thus 10 year flood and 100 year flood only run 140 hours and 240 hours respectively for the purpose of saving running time of FLO-2D model, as explained by RBI (2013). However, both of the time frames are too short for representation of actual floodplain inundation time for 10 year and 100 year flood events. Booth *et al.* (2006) conducted an analysis of flood events on Cosumnes River and estimated the floodplain inundation time based on the peak flow of a flood event and calculated the empirical frequency of each type of flood. According to that study, the floodplain inundation time for a 10 year flood event is estimated to be 18 days (432 hours), and the floodplain inundation time for 100 year flood event is estimated to be 42 days (1008 hours). These estimates are adopted for simulating floodplain recharge in this study.

The 3.5 year flood event floodplain inundation simulation is later extended for additional 6 months right after the 140 hours simulation. In this scenario, it is assumed there is no more inundation water on floodplain, thus groundwater transport happens only between aquifer and non-aquifer materials. This allows the dynamics of groundwater flow in heterogeneous subsurface to be assessed directly.

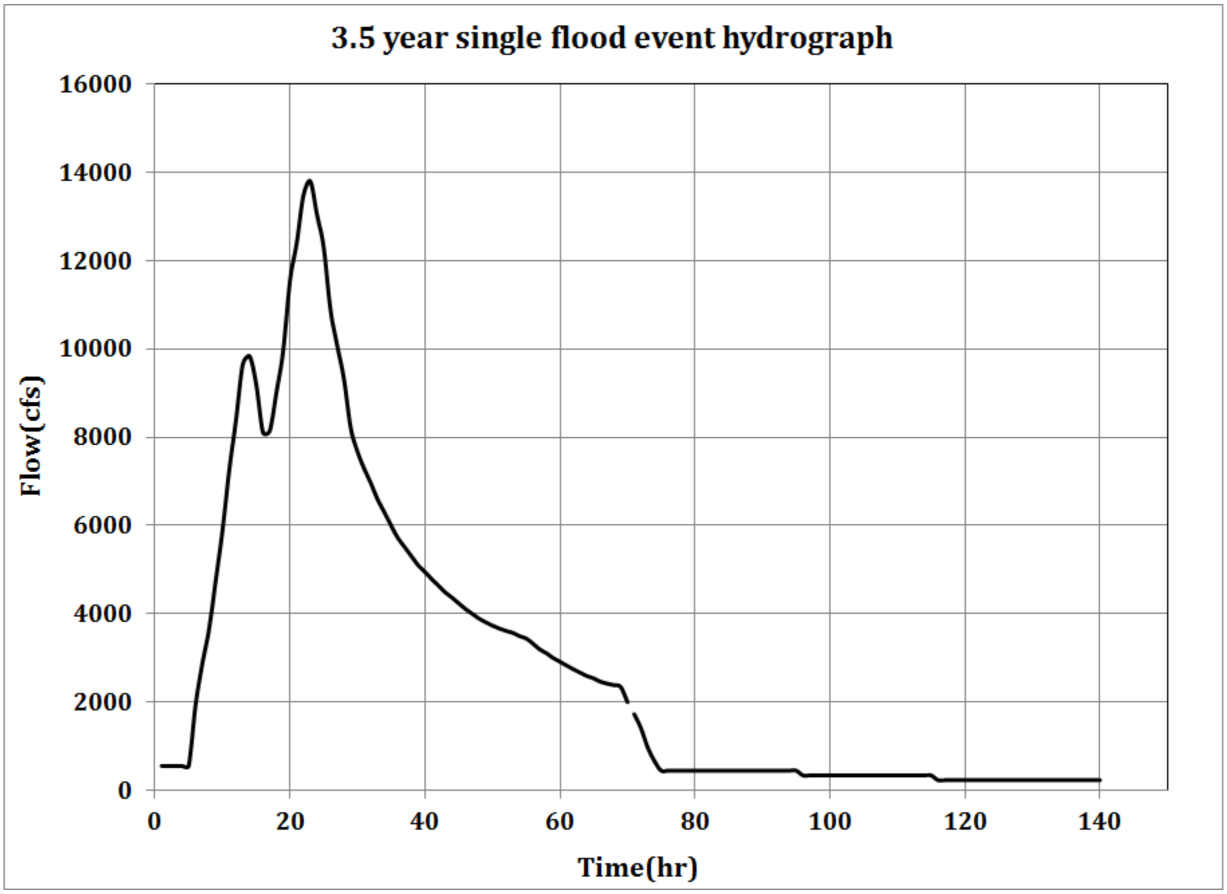


Figure 3.10A 3.5 year single flood event hydrograph.

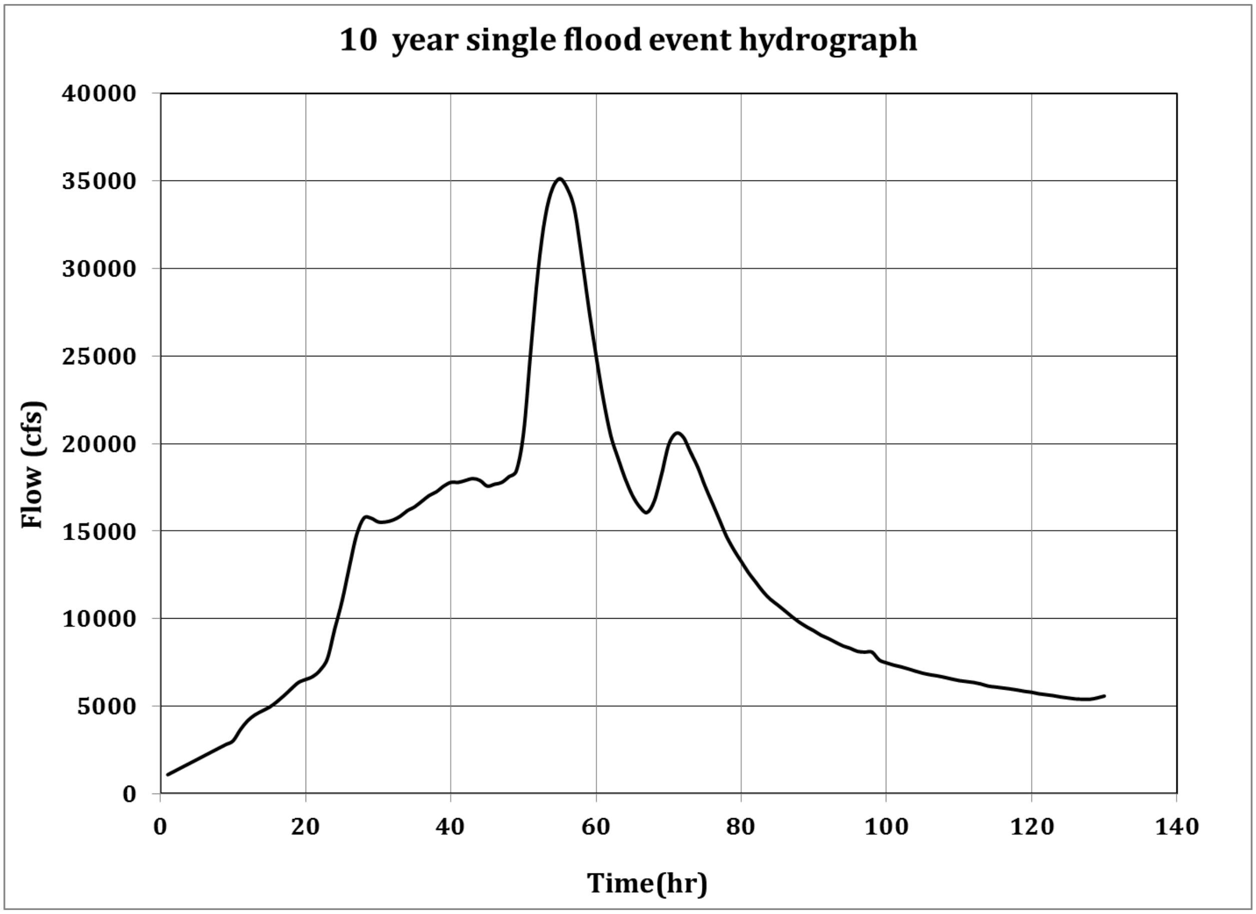


Figure 3.10B 10 year single flood event hydrograph.

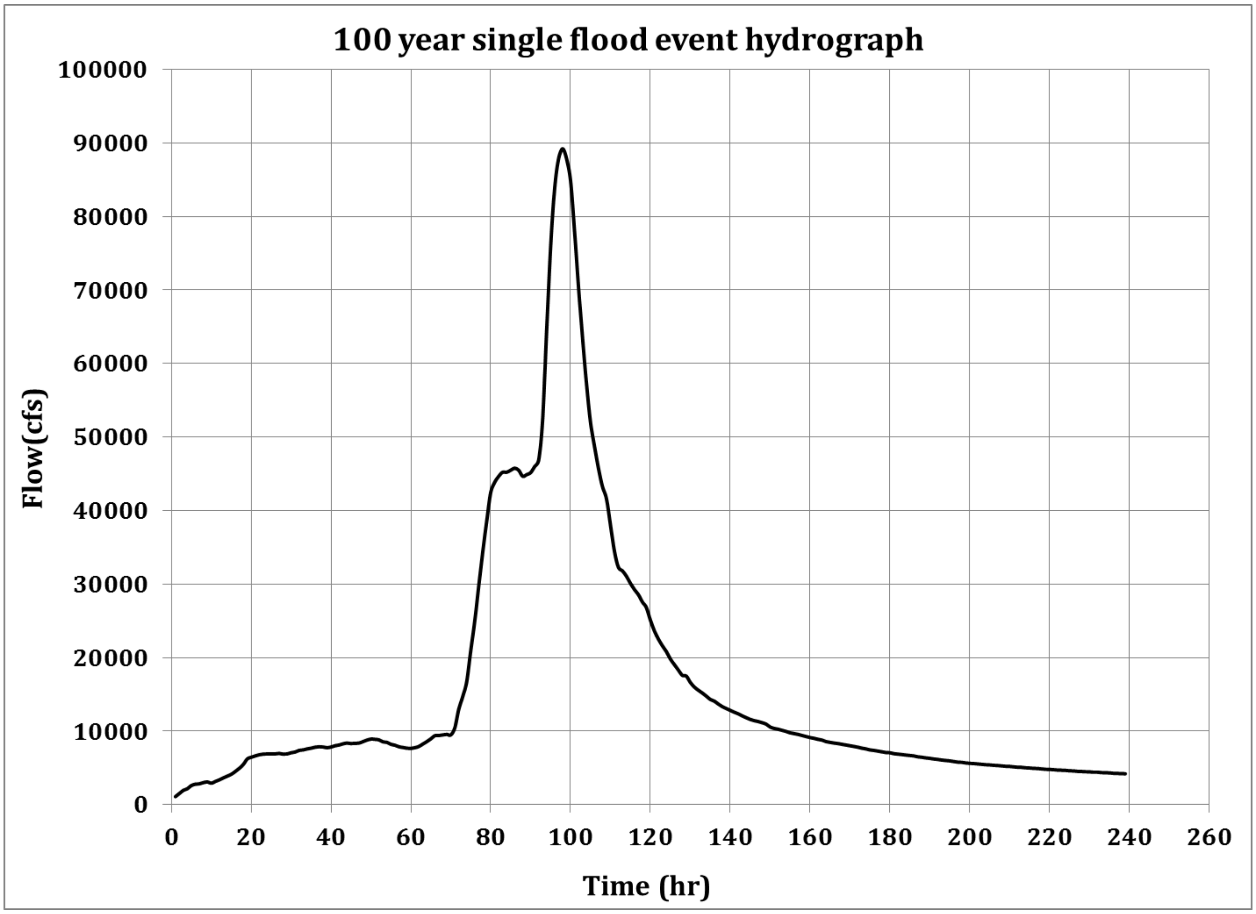


Figure 3.10C 100 year single flood event hydrograph.

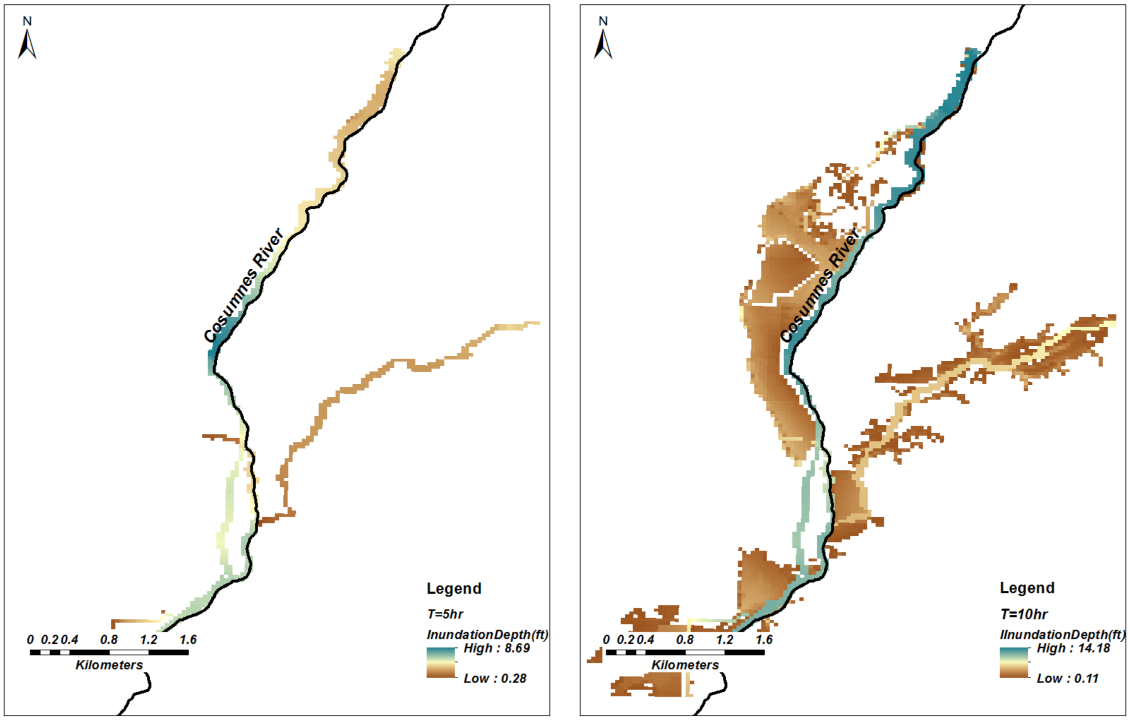


Figure 3.11A 3.5year floodplian inundation map at T=5 hour and T=10 hour.

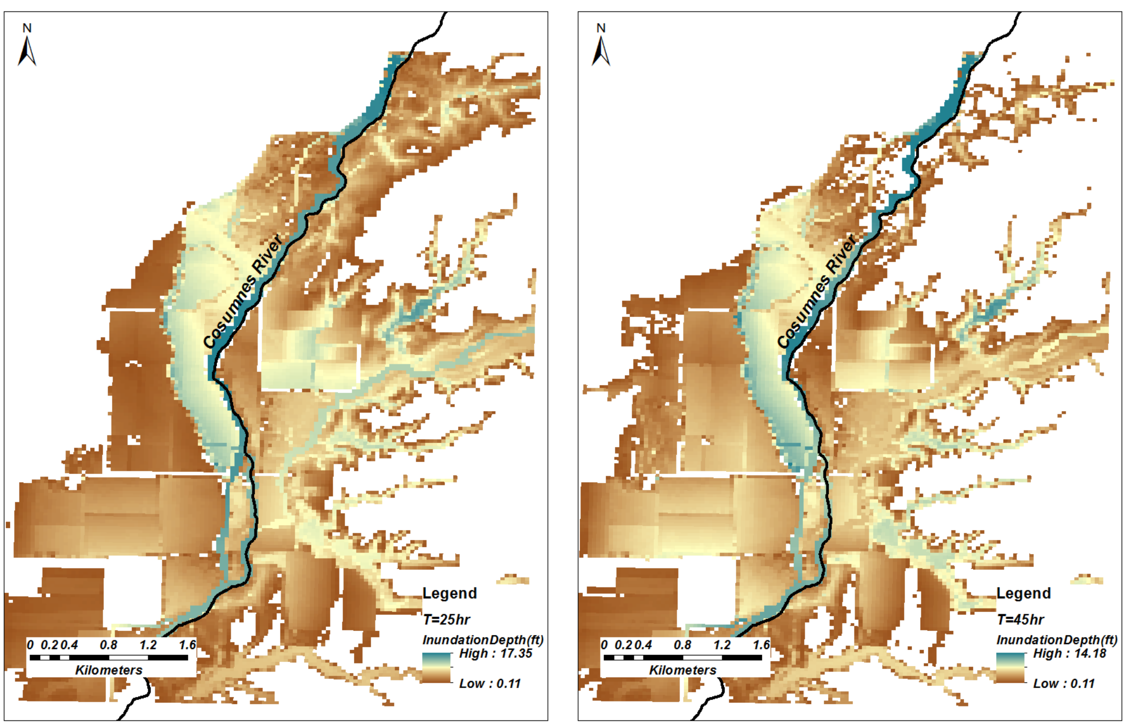


Figure 3.11B 3.5year floodplian inundation map at T=25 hour and T=45 hour.

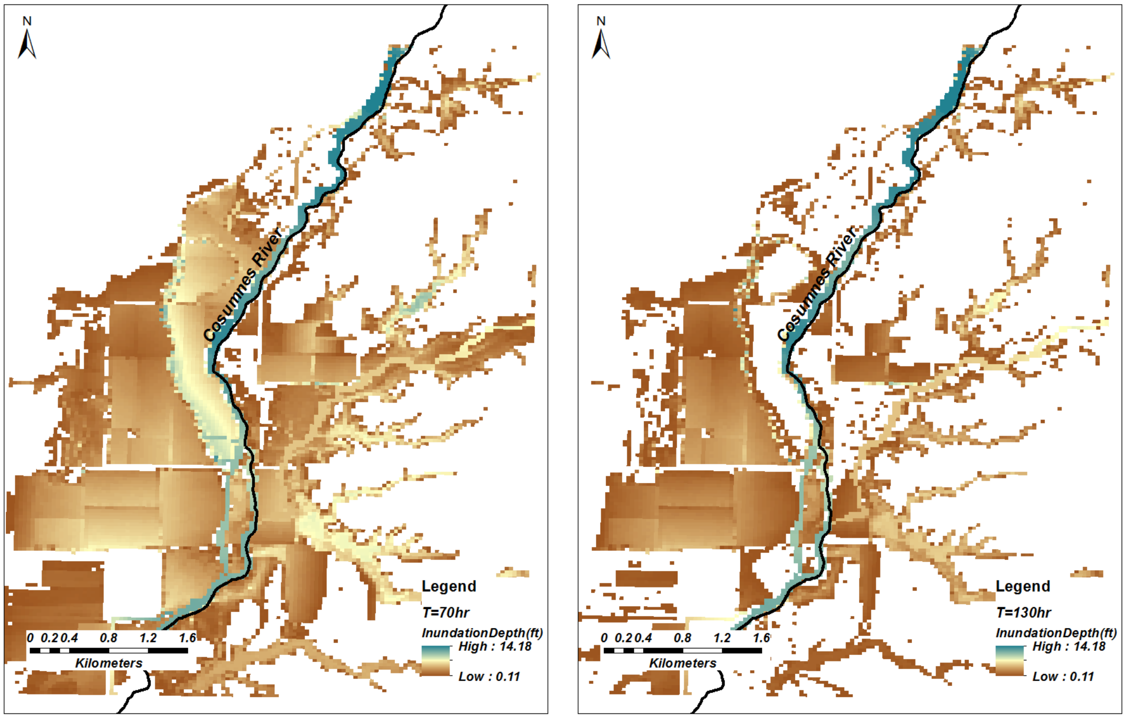


Figure 3.11C 3.5year floodplian inundation map at T=70 hour and T=130 hour.

Chapter 4 Results and discussion

4.1 Model calibration results and discussion

Before conducting the calibration analysis, it should be pointed out that the heterogeneous geological model used in this modeling effort is only one realization of the TPROGS simulation. Therefore, the geological model comes with uncertainties and represents only one possible geological structure among many other possibilities that honors field data. To fully address the uncertainties within the geological model and its impact on model performance, Monte Carlo analysis should be implemented. However, Monte Carlo analysis is a very time consuming process, considering the complexity of the model and calibration time frame. In addition, because the conditioning data, volumetric proportions of the hydro-facies, and regional connectivity of the hydro-facies changes little among all the TPROGS realizations, a full Monte Carlo analysis probably would not produce very different regional responses. Instead, the calibration analysis here focuses on the overall system performance of the flow model.

Recall that the first goal of calibration is to examine whether modeled shallow semi-confined groundwater head response and deep confined groundwater head response are preserved and consistent with field observations. To evaluate this calibration goal and to account for the uncertainties introduced by the geological model as well as to compare the simulated semi-confined and confined response with the field observations, 210 hypothesized monitoring wells are randomly sampled across the model domain. Both sallow semi-confined groundwater head and deep confined groundwater head were extracted. The vertical distance between “semi-confined” and “confined” aquifer is at least 30 meters apart to insure they do not sample the same aquifer. Measured and simulated groundwater head between field observation and the randomly sampled well are compared.

For the deep confined aquifer response, the simulated confined groundwater head in randomly sampled wells are compared with the observed groundwater head data from DWR monitoring wells (Figure D.3), because the DWR monitoring wells virtually measure deep groundwater head. Here the groundwater seasonal variation (e.g. difference between seasonal high and seasonal low) and general head pattern are the comparing variables. If similar groundwater head pattern and seasonal variation is observed in both DWR monitoring well and some random sampled well nearby, it can serve as evidence that the model performance is consistent with field observations. Again, it is highly unlikely that perfect match in both seasonal variation and general head pattern will be achieved because the geological model represents only one possibility among many possible geological structures. Some of the comparison between simulated and observed deep confined groundwater head patterns is presented below (Figure 4.1A, B, C, D, E). One can refer to appendix D for more details.

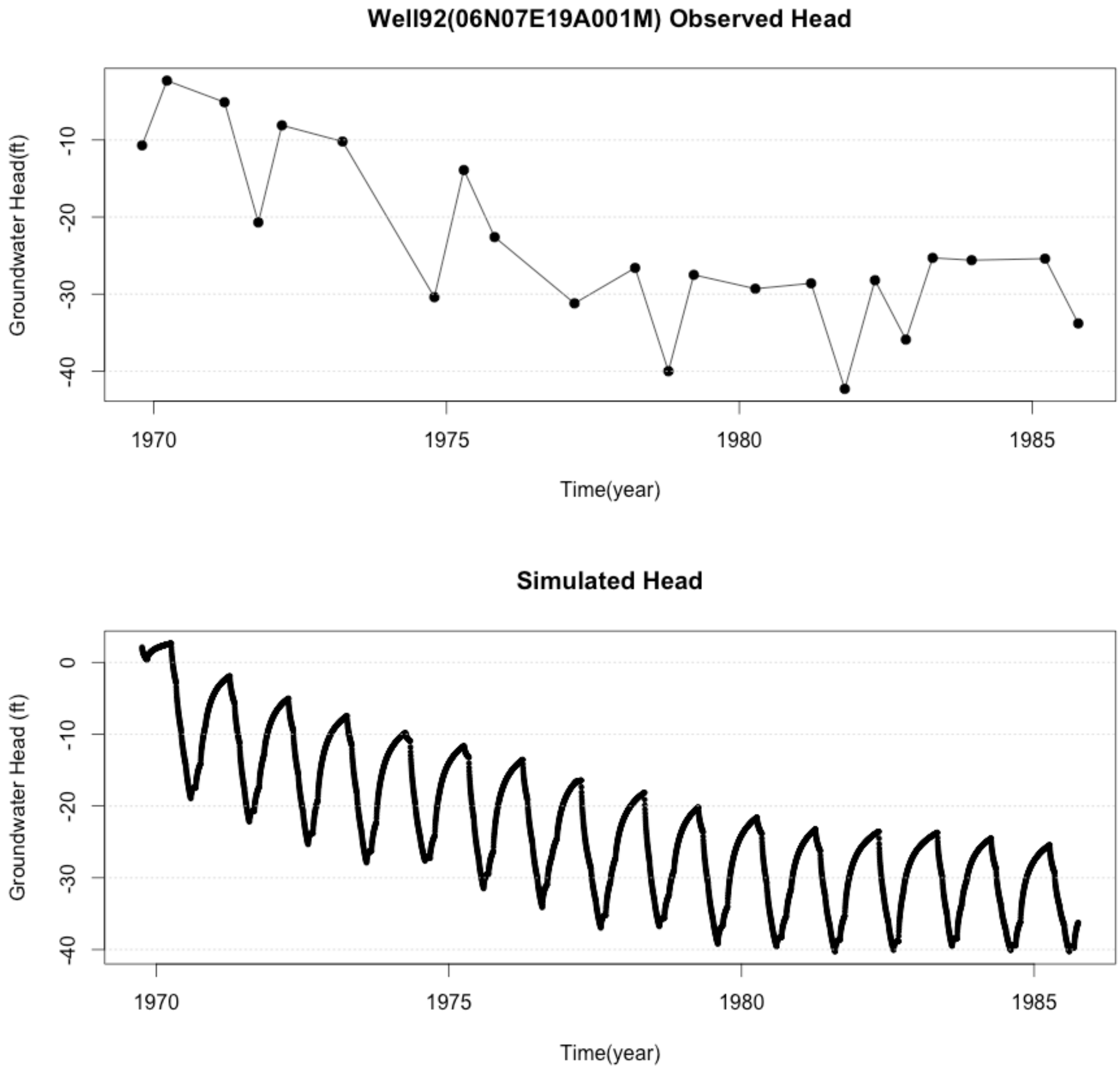
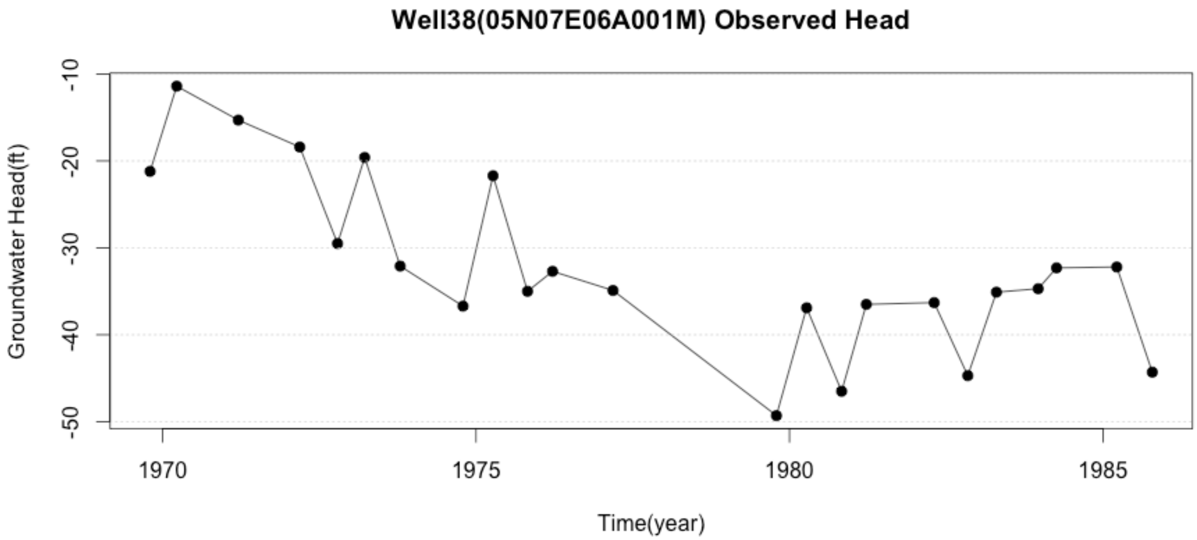


Figure 4.1A Observed and simulated groundwater head response at well 92 (06N07E19A001M).



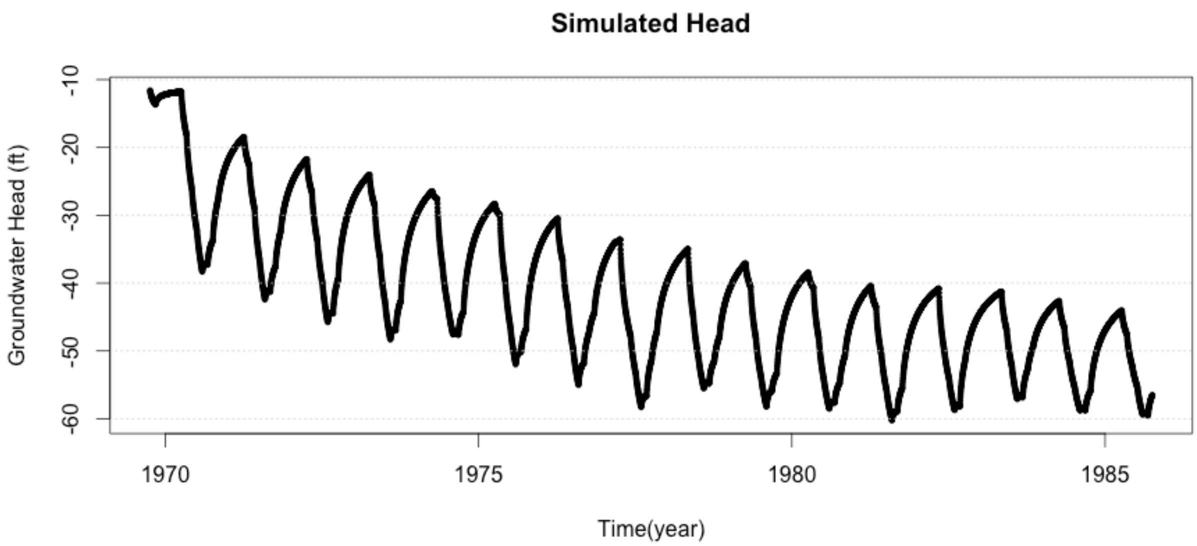


Figure 4.1B Observed and simulated groundwater head response at well38 (05N07E06A001M).

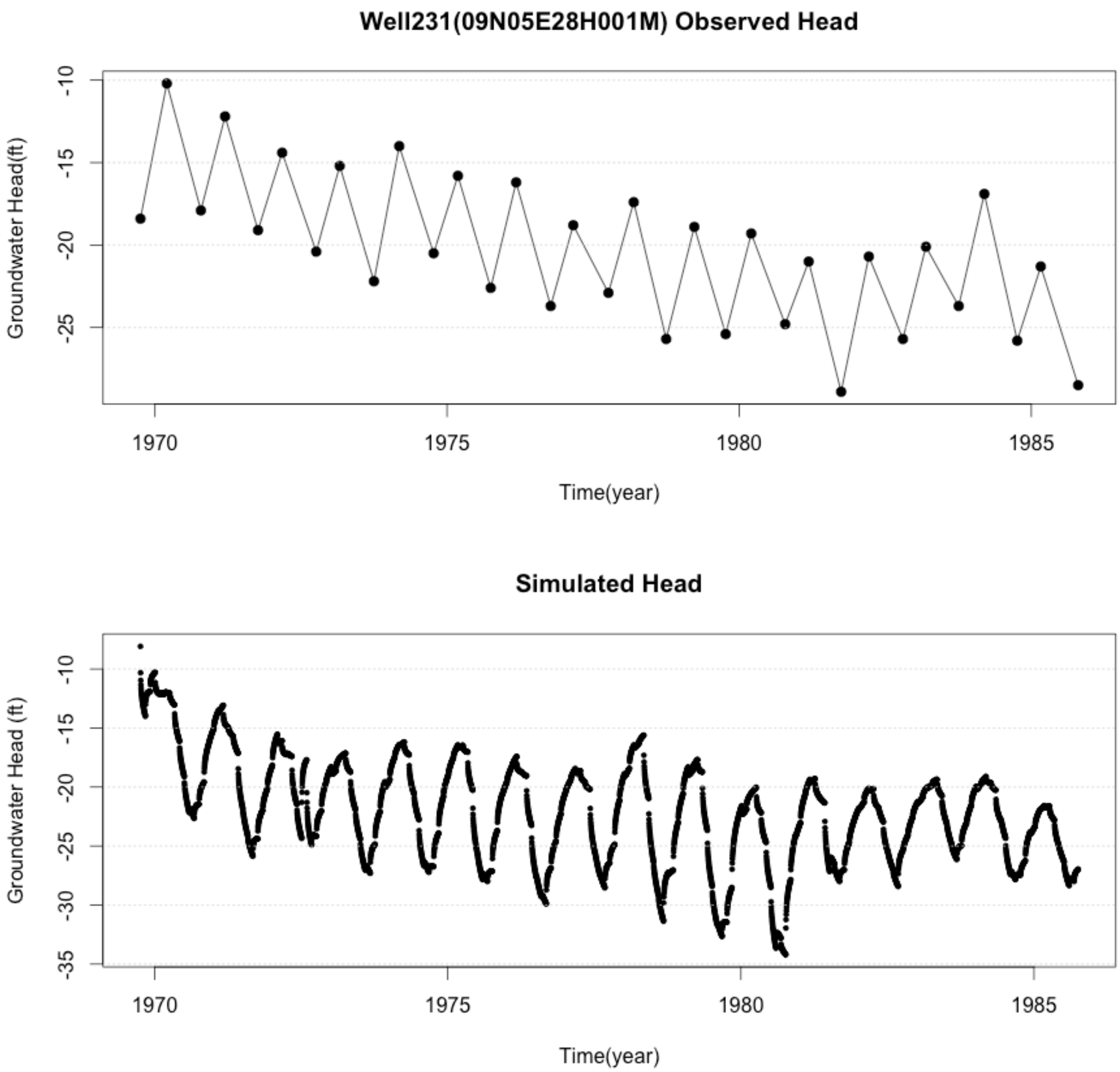


Figure 4.1C Observed and simulated groundwater head response at well231 (09N05E28H001M).

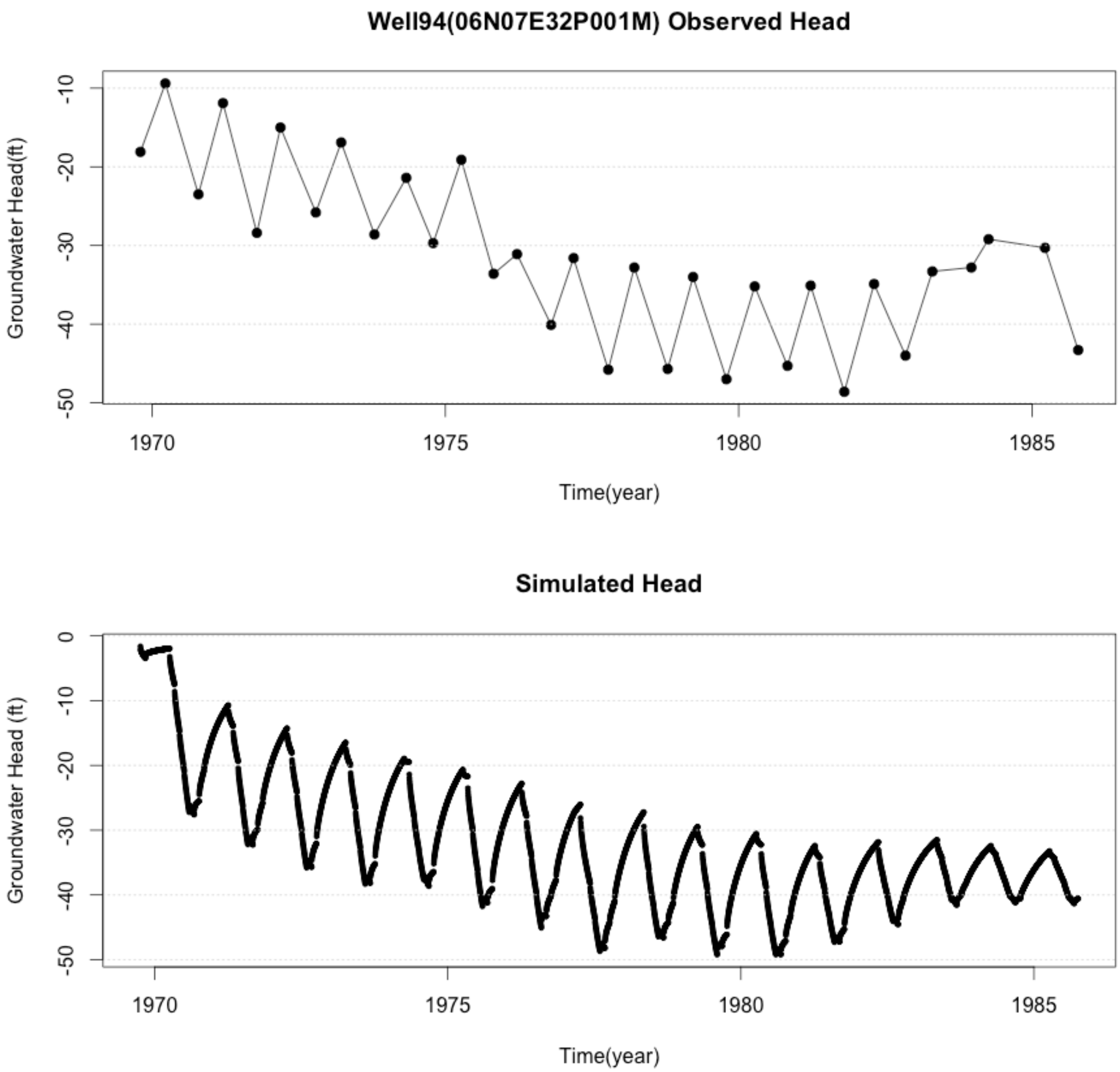


Figure 4.1D Observed and simulated groundwater head response at well94 (06N07E32P001M).

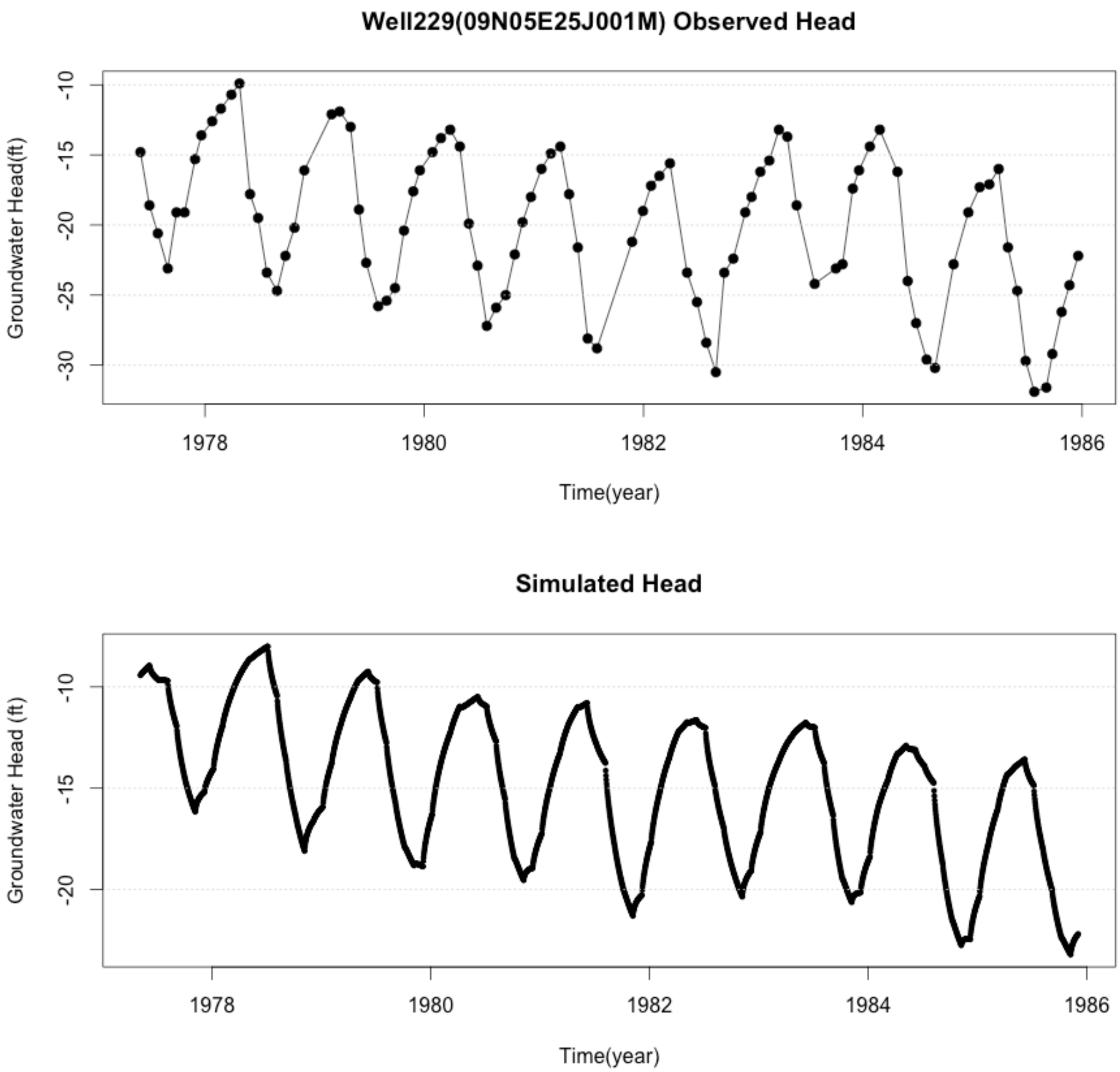


Figure 4.1E Observed and simulated groundwater head response at well229 (09N05E25J001M).

For the shallow semi-confined head response, the simulated semi-confined groundwater head in randomly sampled wells are compared with field observations in shallow monitoring wells (Cosumnes River Group (CRG) monitoring wells) at the Cosumnes River study site (Figure D.3). However, there are no historical measurements of shallow groundwater head within calibration period (1969-1985). Available data is from 2000 to 2011 (CRG UCD wells) and December 2012 to August 2014 (CRG MW wells). Recall the calibration goal here is to examine whether the shallow moderate groundwater head response is preserved in the model or not, thus field observation of shallow groundwater response in CRG wells can still provide information on how the shallow response should be, even though the observed data do not fall into the calibration period. Here the groundwater seasonal variation (e.g. difference between seasonal high and seasonal low) is the comparing variable. Comparison schemes are the same as the deep responses comparing process outlined in previous paragraph. Some of comparing results are shown in Figure 4.2A, B, C, D. One can refer to appendix D for more details.

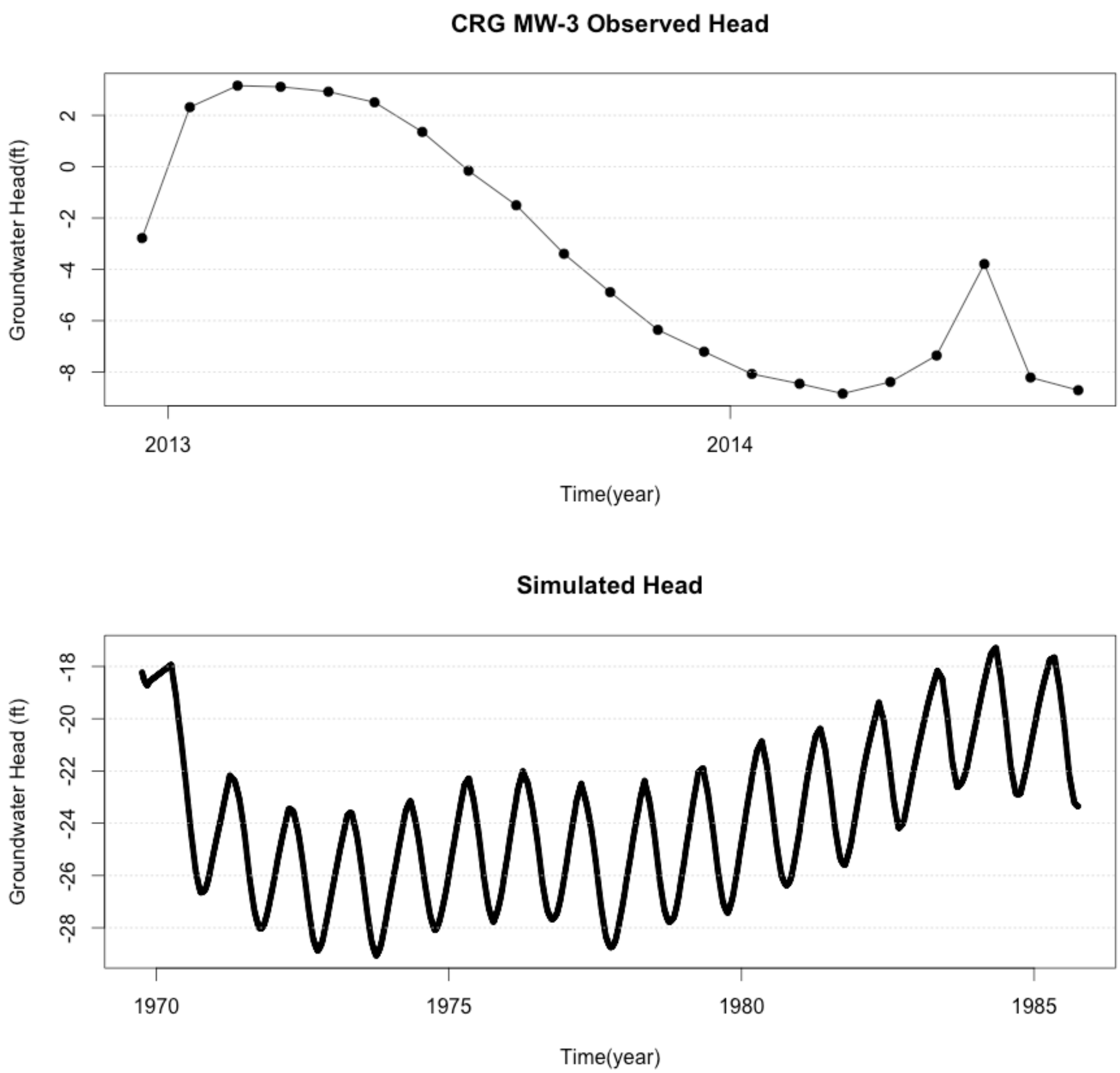


Figure 4.2A Observed and simulated groundwater head response at well CRG MW-2.

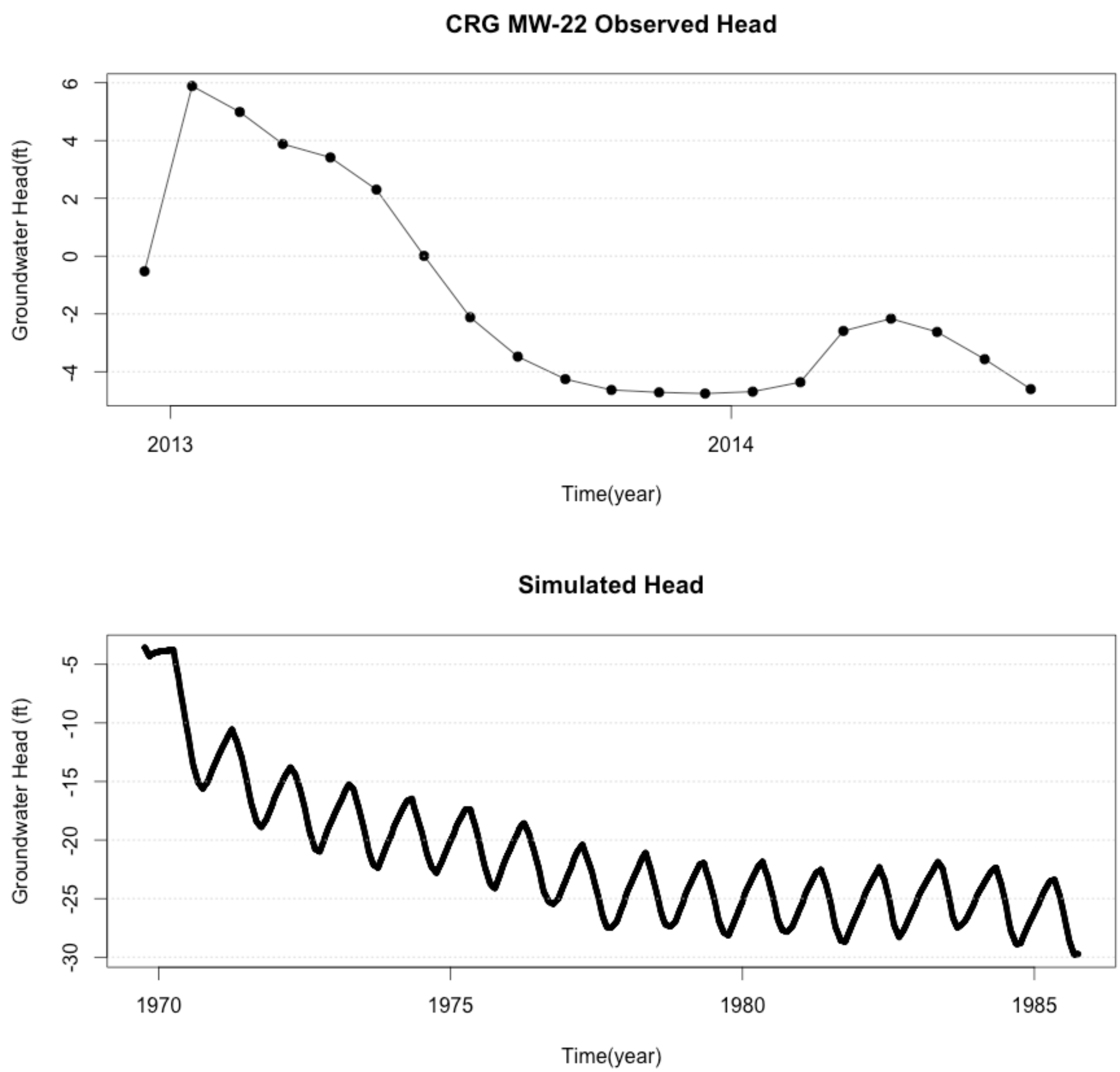


Figure 4.2B Observed and simulated groundwater head response at well CRG MW-22.

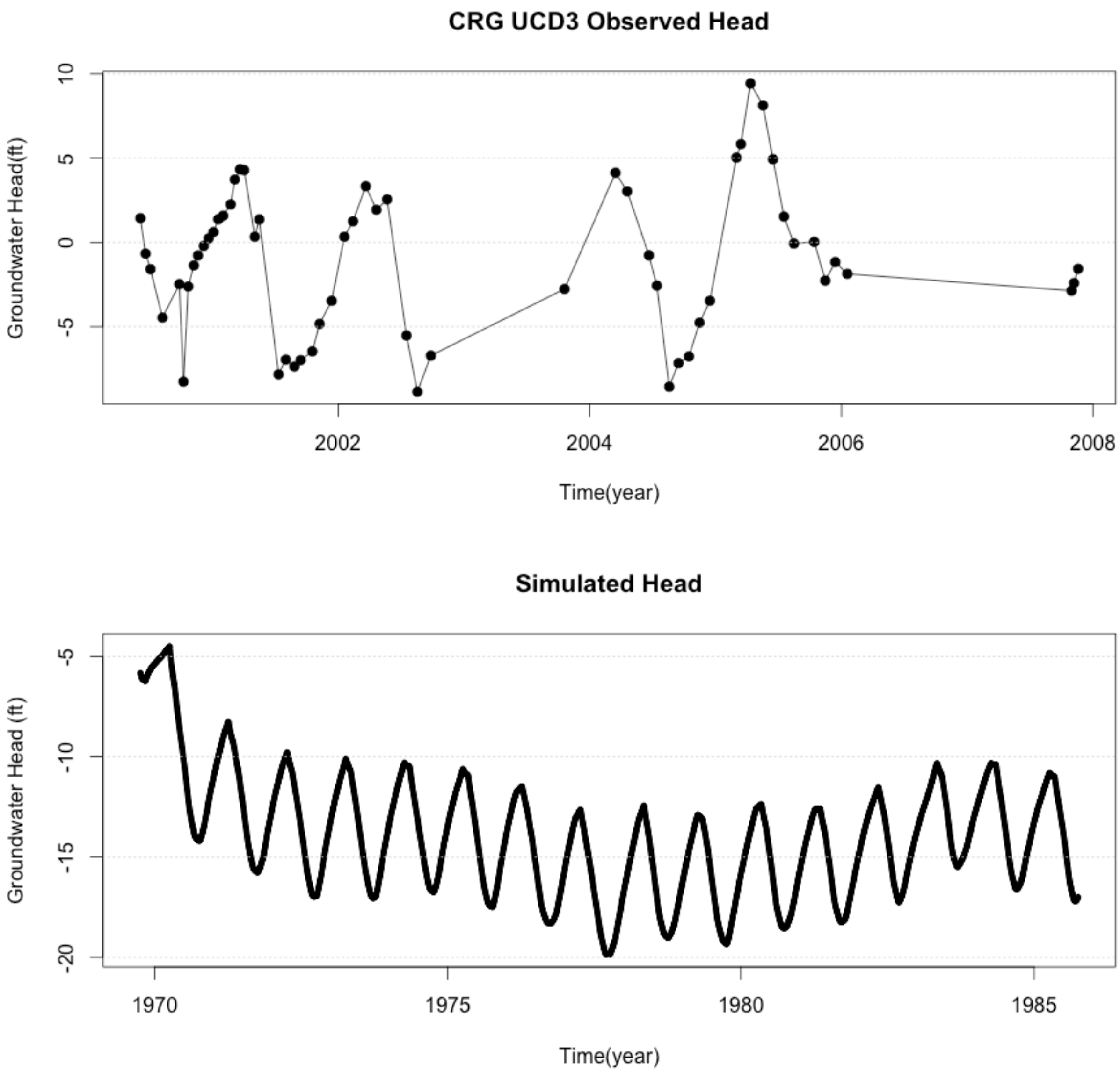
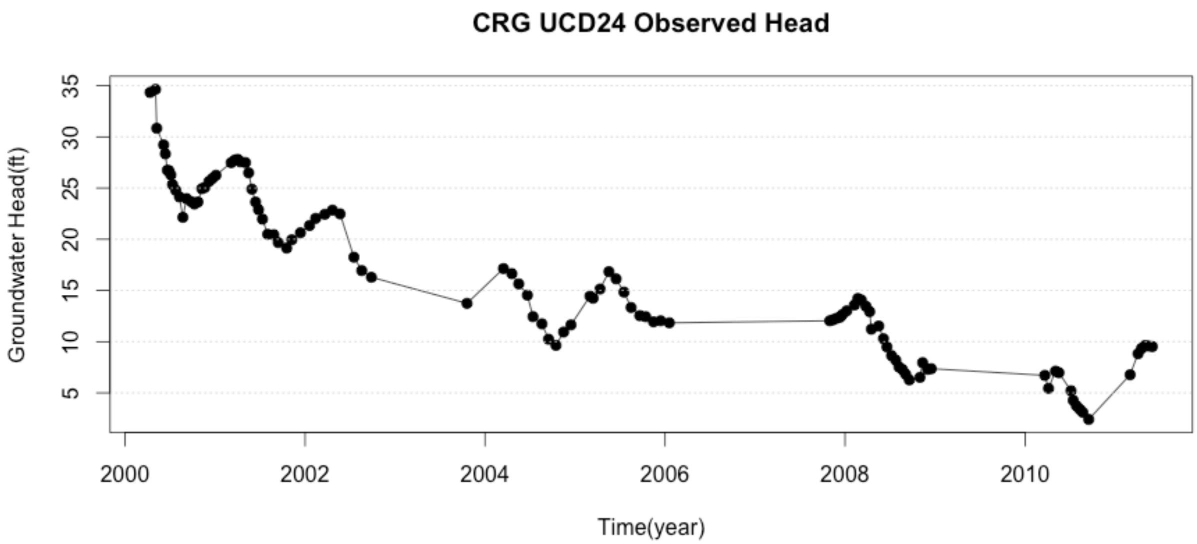


Figure 4.2C Observed and simulated groundwater head response at well CRG UCD3.



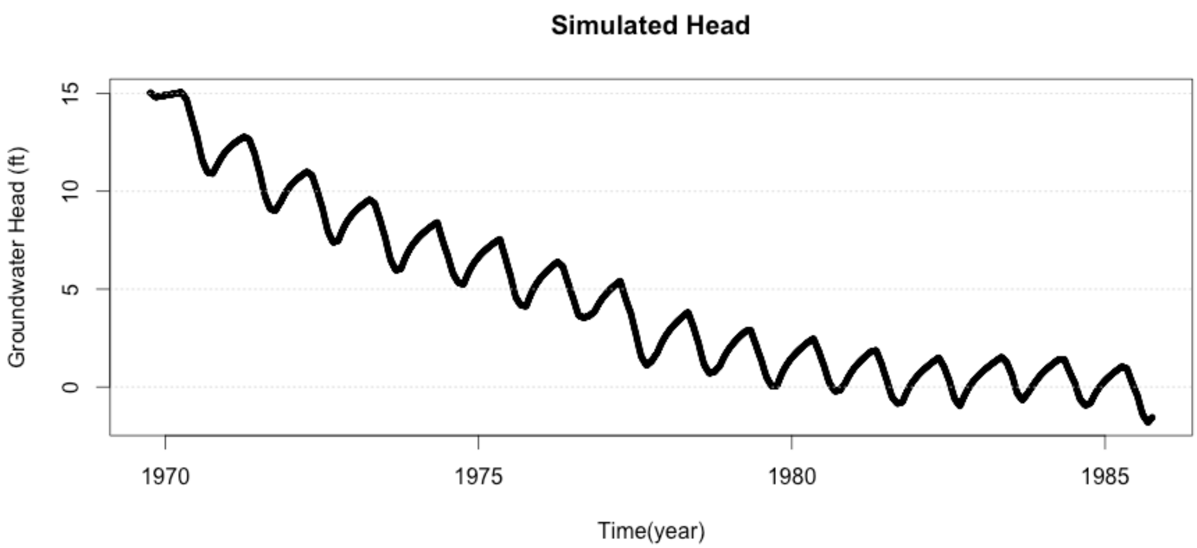


Figure 4.2D Observed and simulated groundwater head response at well CRG UCD24.

Figure 4.1A-E, the simulated and observed deep confined groundwater head response generally shows good match both in seasonal variation and general patterns. The deep confined groundwater head seasonal variations are generally larger than 15 feet. Shown in Figure 4.2A-D, the simulated shallow semi-confined groundwater head seasonal variations generally are on the order of 10 feet or less than 10 feet and are moderated. Comparing the simulated shallow semi-confined seasonal variation and the simulated deep confined seasonal variation, it is seen that the shallow response is more gentle than the deep groundwater response, which means the shallow to deep groundwater head response is preserved in the model. Therefore the overall model response is consistent with the field observations.

To evaluate the second calibration goal: whether simulated groundwater head matches with observed groundwater head at selected calibration wells. 39 monitor wells among the 270 DWR monitoring wells that fall in model domain are chosen as representative calibration wells (Figure D.3). The chosen wells spread evenly on the model domain and most of them have observed groundwater head data in the calibration period (1969-1985). The simulated groundwater head are compared with the observed groundwater head for each of the representative calibration wells. Again, because the geological model used in this study is only one realization of the TPROGS simulation, which represents only one possible geological structure that honors data. It is highly unlikely the simulated groundwater head will match with all observed head in specific DWR monitoring wells. Below are some of the calibration results (Figure 4.3A, B, C, D, E). One can refer to appendix D for more details.

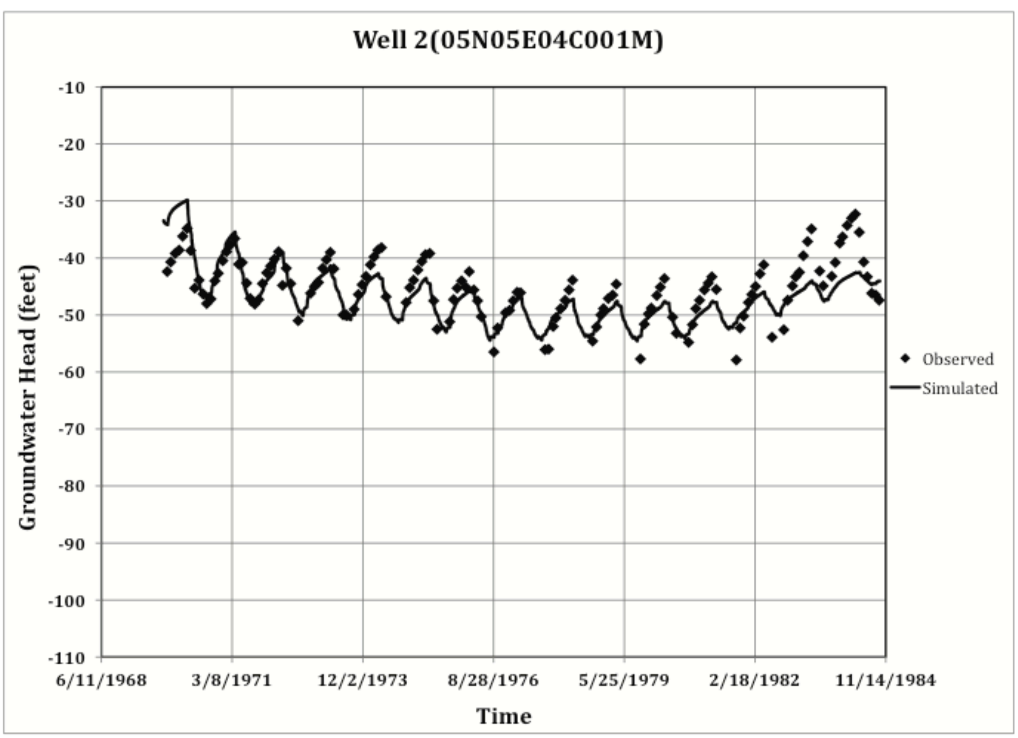


Figure 4.3A Observed and simulated groundwater head response at well2 (05N05E04C001M).

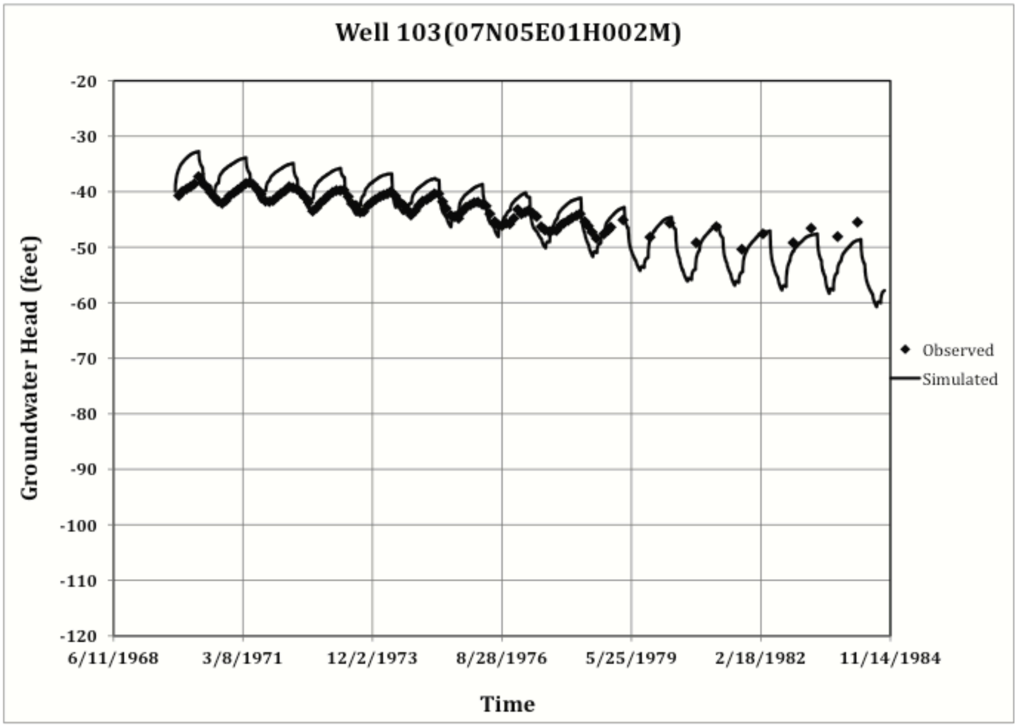


Figure 4.3B Observed and simulated groundwater head response at well103 (07N05E01H002M).

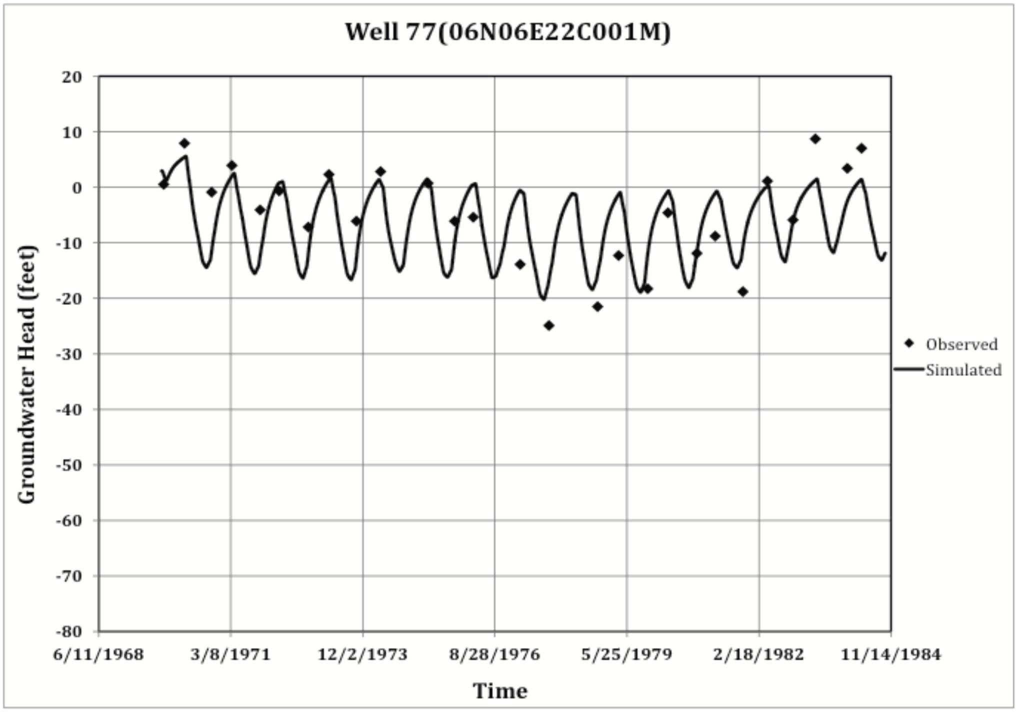


Figure 4.3C Observed and simulated groundwater head response at well77 (06N06E22C001M).

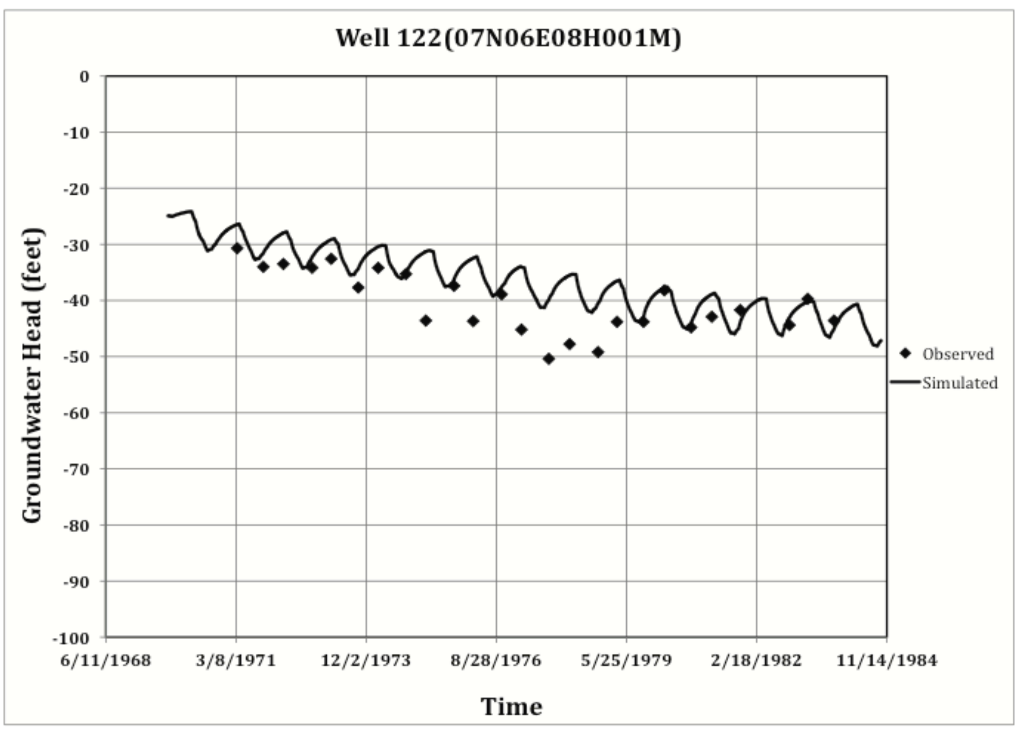


Figure 4.3D Observed and simulated groundwater head response at well122 (07N06E08H001M).

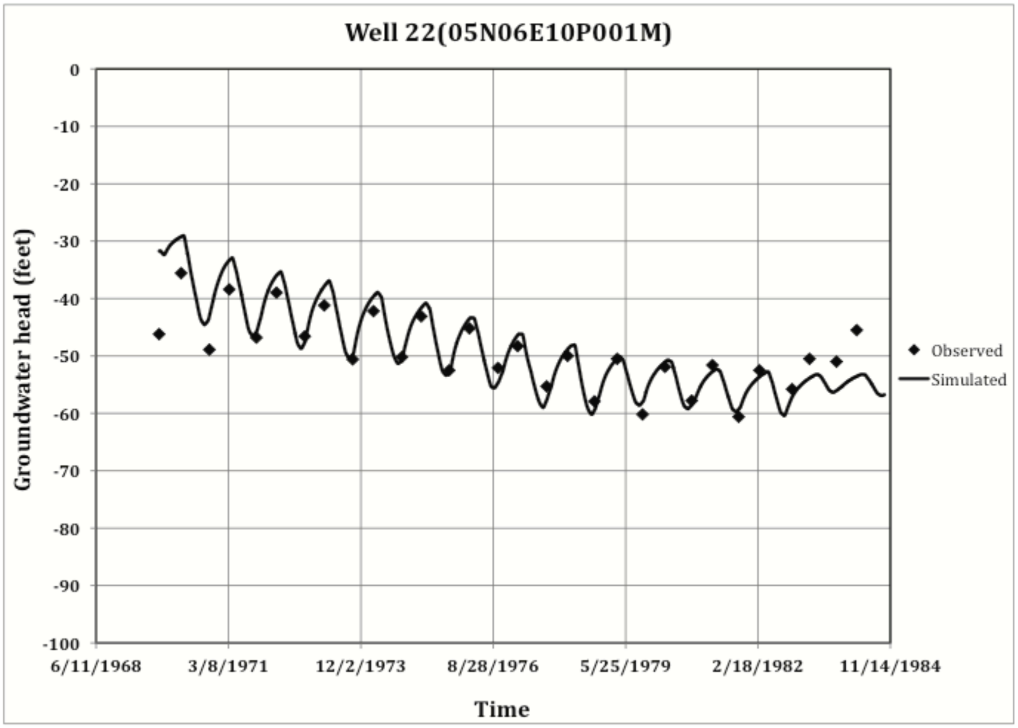


Figure 4.3E Observed and simulated groundwater head response at well22 (05N06E10P001M).

Residuals between observed groundwater head and corresponding simulated groundwater head at selected calibration wells are calculated and analyzed. In the residual analysis, the goodness of fit statistical variables are calculated, namely goodness of fit coefficient and root mean square error. Root mean square error (RMSE) is a measurement for the averaged difference between simulated groundwater head and observed groundwater head (residuals). It is defined mathematically as equation (15),

$$RMSE = \sqrt{E((h_{observed} - h_{simulated})^2)} \qquad (15)$$

In this calibration analysis, the square root mean error of residuals is 5.33 feet, which means the simulated head falls within 5.33 feet of observed head on average for selected calibration wells.

The goodness of fit coefficient provides a measurement of how well the simulated head fits with the observed head. It is defined mathematically as equation (16),

$$R = \frac{\sum((h_{observed} - m_{observed})(h_{simulated} - m_{simulated}))}{\left(\sum(h_{observed} - m_{observed})^2 \sum(h_{simulated} - m_{simulated})^2\right)^{0.5}} \qquad (16)$$

The goodness of fit coefficient is 0.9708, indicating a very good fit between simulated groundwater head and observed groundwater head (Figure 4.4). The histogram of residuals (Figure 4.5) shows that 70 percent of the simulated groundwater head are within 5 feet of observed value, 94.46 percent of the simulated groundwater head are within 10 feet of the observed value, for the selected calibration wells.

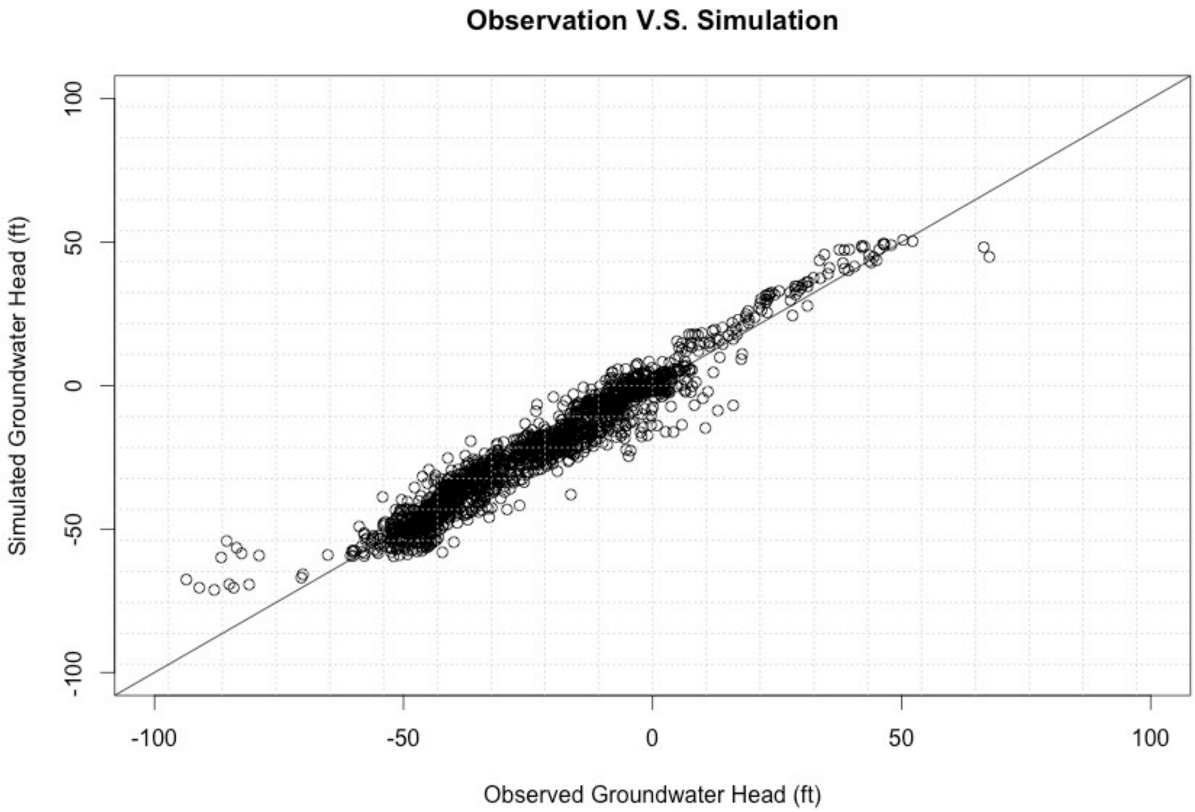


Figure 4.4 Observed groundwater head and the simulated groundwater head for selected monitoring wells.

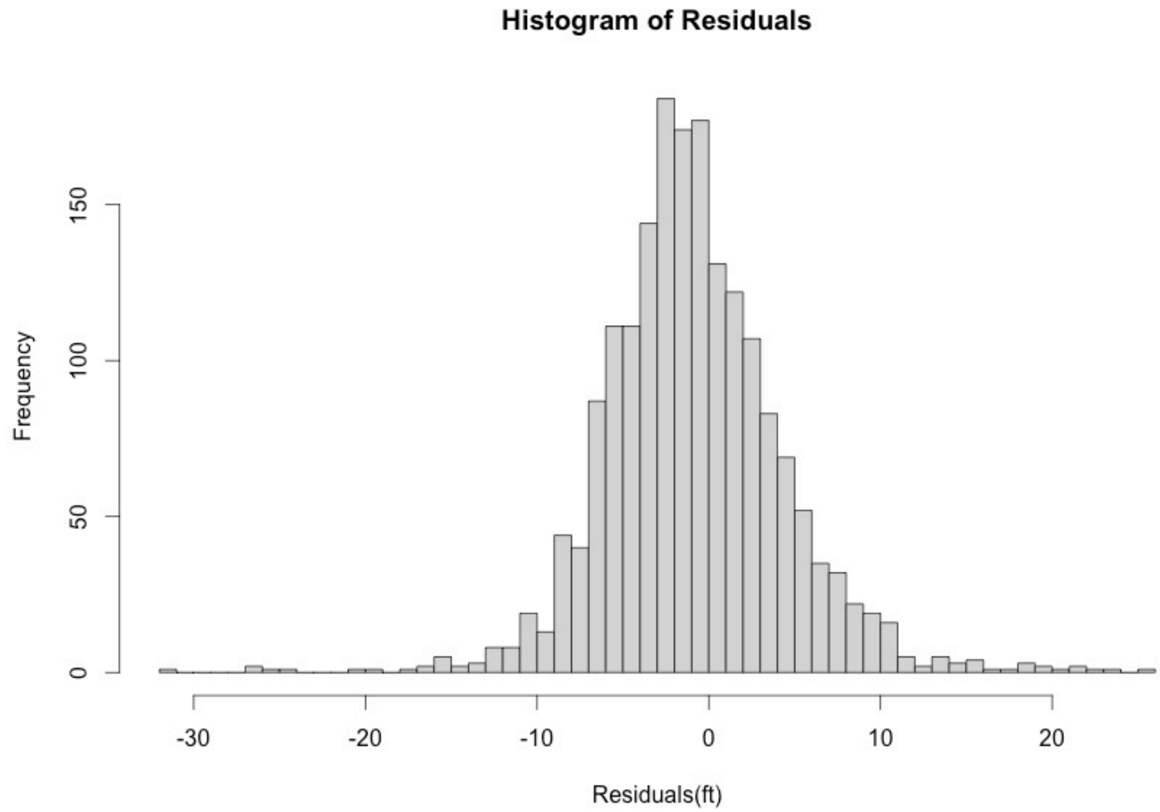


Figure 4.5 Histogram of groundwater head residual distribution for selected monitoring wells.

In summary, calibration analysis above suggest that the overall model performance is consistent with field observations. And the simulated groundwater head matches well with the observed groundwater head at selected calibration wells. The initial estimates of parameter values and final calibrated parameter values are listed in Table 4.1 and Table 4.2, respectively.

To achieve the calibration goal, the pumping for agricultural region is also adjusted when necessary. The percentage of adjustment in pumping for agricultural region is shown in Figure 4.6. The percentage of adjustment is calculated by equation (17)

$$Percentage = \frac{adjusted\ pumping - original\ pumping}{original\ pumping} \qquad (17)$$

Where *adjusted pumping* represents pumping applied in this model, while *original pumping* represents pumping from SACIWRM model.

The maximum adjustment is less than 15 percent, that falls in the error bounds (typically less than 20-50 percent) of estimating groundwater pumping using conventional crop consumptive use method, which was implemented in SACIWRM (RMC 2011) model. Therefore, the pumping applied in this model is reasonable.

Table 4.1 The initial estimate of hydraulic parameters.

Hydro-Facies	Hydraulic Conductivity	Specific Storage	Porosity	VG-Alpha	VG-N	VG-Residual Saturation	VG-Saturated Saturation
Gravel	45.26 m/day	2.0e-5	0.35	3.55	3.16	0.1	1.0
Sand	27.12m/day	5.0e-5	0.35	3.55	3.16	0.1	1.0
Muddy Sand	0.1m/day	1.0e-4	0.4	2.69	2.0	0.1	1.0
Mud	0.001m/day	1.0e-3	0.45	1.62	2.0	0.2	1.0
Deep Aquifer	83.17m/day	4.8e-4	0.35	3.55	3.16	0.1	1.0

Table 4.2 The final calibrated parameter values.

Hydro-Facies	Hydraulic Conductivity	Specific Storage	Porosity	VG-Alpha	VG-N	VG-Residual Saturation	VG-Saturated Saturation
Gravel	67.52 m/day	4.0e-5	0.35	3.55	3.16	0.1	1.0
Sand	41.24 m/day	8.0e-5	0.35	3.55	3.16	0.1	1.0
Muddy Sand	0.2m/day	1.0e-4	0.4	2.69	2.0	0.1	1.0

Mud	0.0017m/day	1.0e-3	0.45	1.62	2.0	0.2	1.0
Deep Aquifer	45m/day	1.0e-2	0.6	3.55	3.16	0.1	1.0

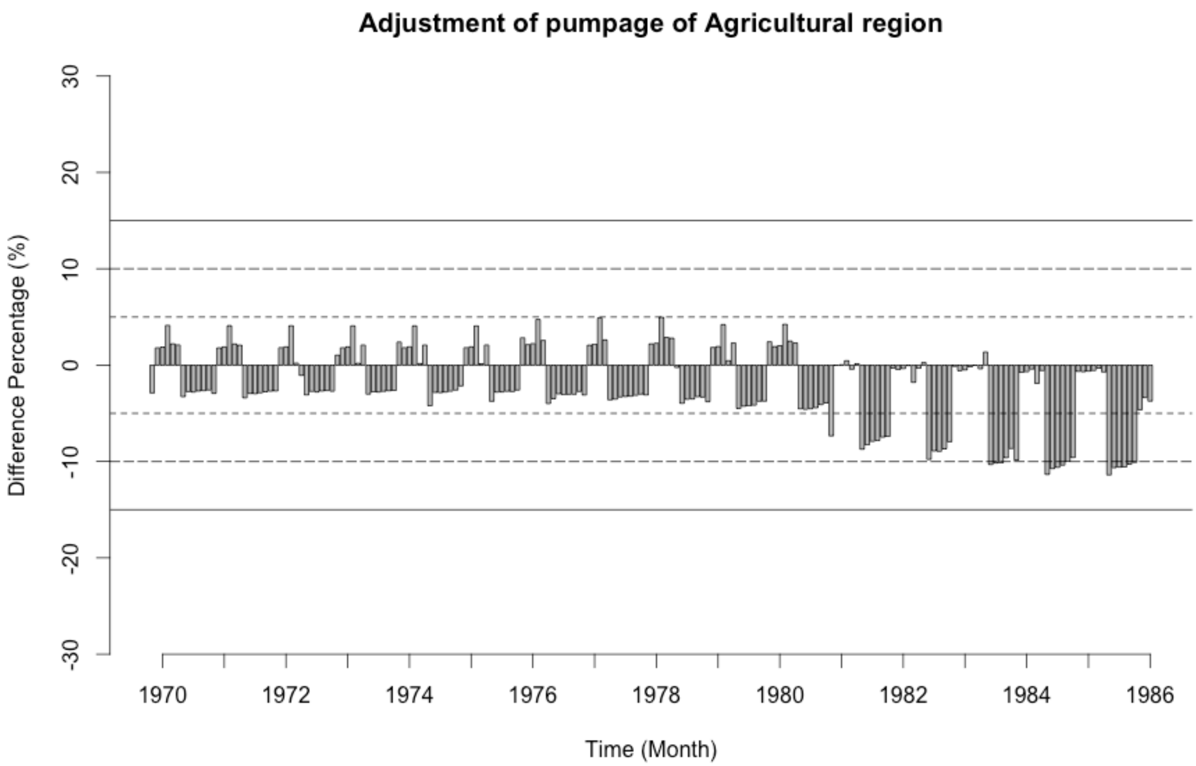


Figure 4.6 The percentage of adjustment in pumping for agricultural region.

4.2 Floodplain inundation simulation results and discussion

4.2.1 Subsurface storage change of the four scenarios

Recall the four floodplain inundation scenarios listed in Table 3.4. For each of the scenarios, cumulative subsurface storage change in floodplain aquifer materials (gravel and sand), floodplain non-aquifer materials (muddy sand and mud) and entire floodplain subsurface are calculated and plotted in Figure 4.7A, B, C and listed in Table 4.3.

Similar storage change patterns are observed in all 3.5, 10, 100 year flood events as shown in Figure 4.7A, 4.7B and 4.7C. The cumulative storage change is increasing in both aquifer material and non-aquifer materials, but the storage change in the aquifer materials is much larger than the storage change in non-aquifer materials since the beginning of floodplain inundation. Correspondingly, the storage changing rate (Figure 4.8 for 3.5 year flood event) for aquifer materials is much larger than that of non-aquifer materials, especially at the early inundation time. Before reaching flood peak, storage change in both aquifer and non-aquifer materials are quicker and at a larger magnitude than after flood peak passed as shown in Figure 4.8. The rate of change drops after flood peak for both aquifer and non-aquifer materials. The entire recharge and storage change process is controlled by the hydraulic conductivity, storage capacity of aquifer, non-aquifer materials as well as the flooding dynamics.

When Cosumnes River basin receives intense rainfall, the river discharge goes up quickly, and floodplain at lower Cosumnes River (this study site) starts flooding when river discharge at Michigan Bar reaches 800 cubic feet per second (Personal Communication 2014, Andrew Nichols). Before reaching peak flood, the flooding water expands, covering more and more floodplain surface area. Correspondingly, the depth of inundation water gets deeper. After the flood peak passes, river discharge recedes and the flood water starts flowing back into the Cosumnes River. As a result, the flooding surface area shrinks and inundation water depth becomes shallower.

In the process of flooding water expanding, recharge happens very quick due to the high hydraulic gradient between surface flood water and the dry subsurface in both aquifer sediments and non-aquifer sediments. The larger and quicker storage change in aquifer material but slower and smaller storage change in non-aquifer materials is mainly caused by the difference in hydraulic conductivity of aquifer and non-aquifer materials. In this study, hydraulic conductivity of aquifer material are gravel 67.52 m/day, sand 41.24 m/day, non-aquifer material are muddy sand 0.2 m/day, mud 0.0017 m/day. After flood peak passed, in the process of flooding water back flow to river, surface water depth is decreasing, which results in decreased hydraulic gradient between shallow surface water and near surface groundwater. Thus the storage change in both aquifer material and non-aquifer materials are slower and at a lower rate. However, as shown in Figure 4.8, the rate of storage change of non-aquifer materials is actually larger than the rate of change in aquifer materials at later time of inundation. This is mainly due to the difference in storage capacity of aquifer and non-aquifer materials. Shown in Table 3.5, the volumetric fraction of aquifer materials of floodplain is about 25 percent, while non-aquifers consist of around 75 percent. The relative small volumetric fraction of aquifer-materials limits its storage capacity. Once they are filled up in early time, little un-used storage is available for the rest of flooding period, resulting in small storage change in later time even though their high hydraulic conductivity favors recharge. On the contrary, the non-aquifer materials have large storage capacity. But due to their low hydraulic conductivity, the space is filling up slowly, eventually leading to higher rate of storage change (as compared to aquifer materials).

The stairstep-like shape seen in some of the pre-peak portions of the aquifer storage cumulative change curves is due to the difference in model time step and the time step of inundation map applied (e.g. the model time step is 1.5 hours in this study, while the floodplain inundation map applied changes every 5 or 10 hours). With large time steps in inundation map, the dynamics of inundating cannot be resolved because the flooding water spreads very quick laterally before reaching flood peak. Future work on applying hourly inundation maps can achieve more significant results.

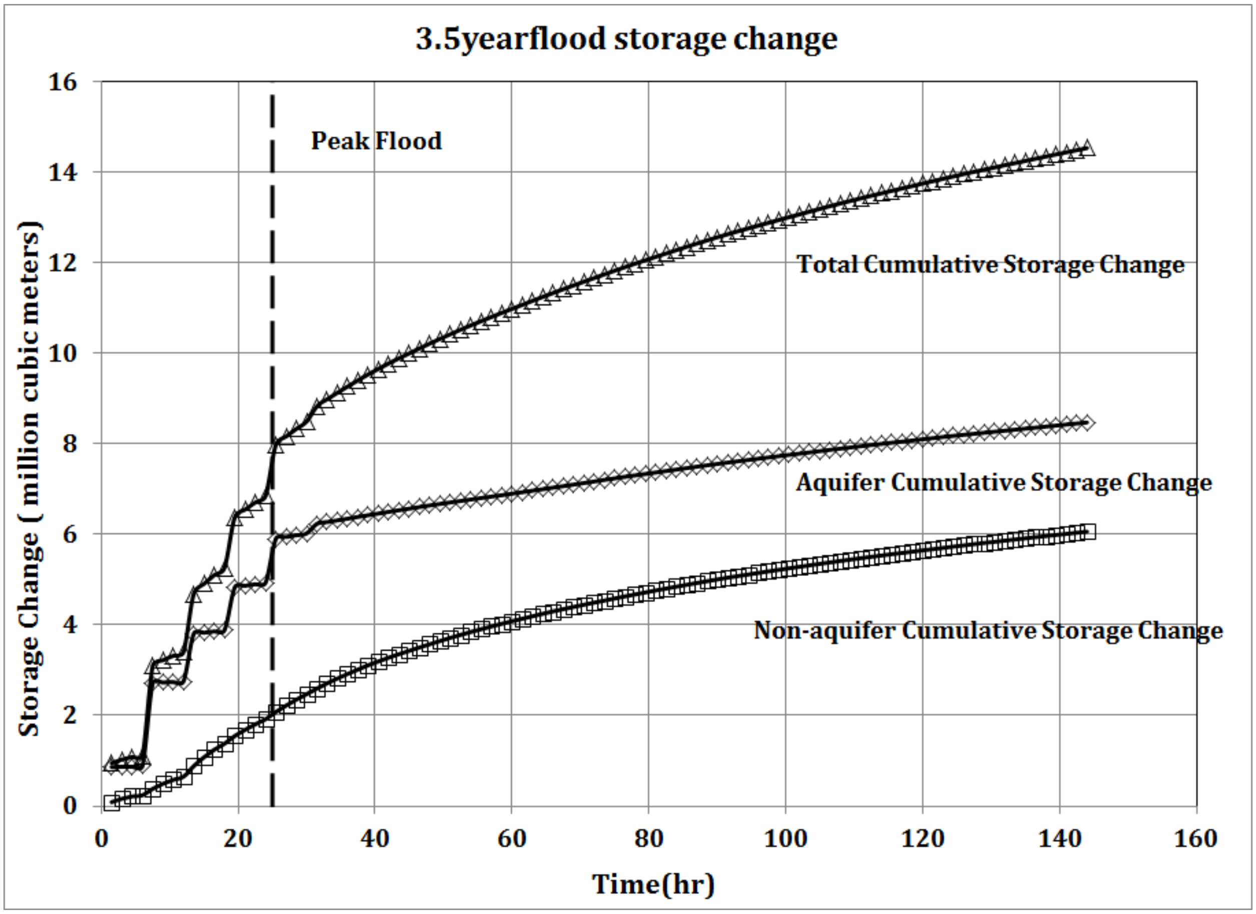


Figure 4.7A Subsurface cumulative storage change for 3.5 year floodplain inundation.

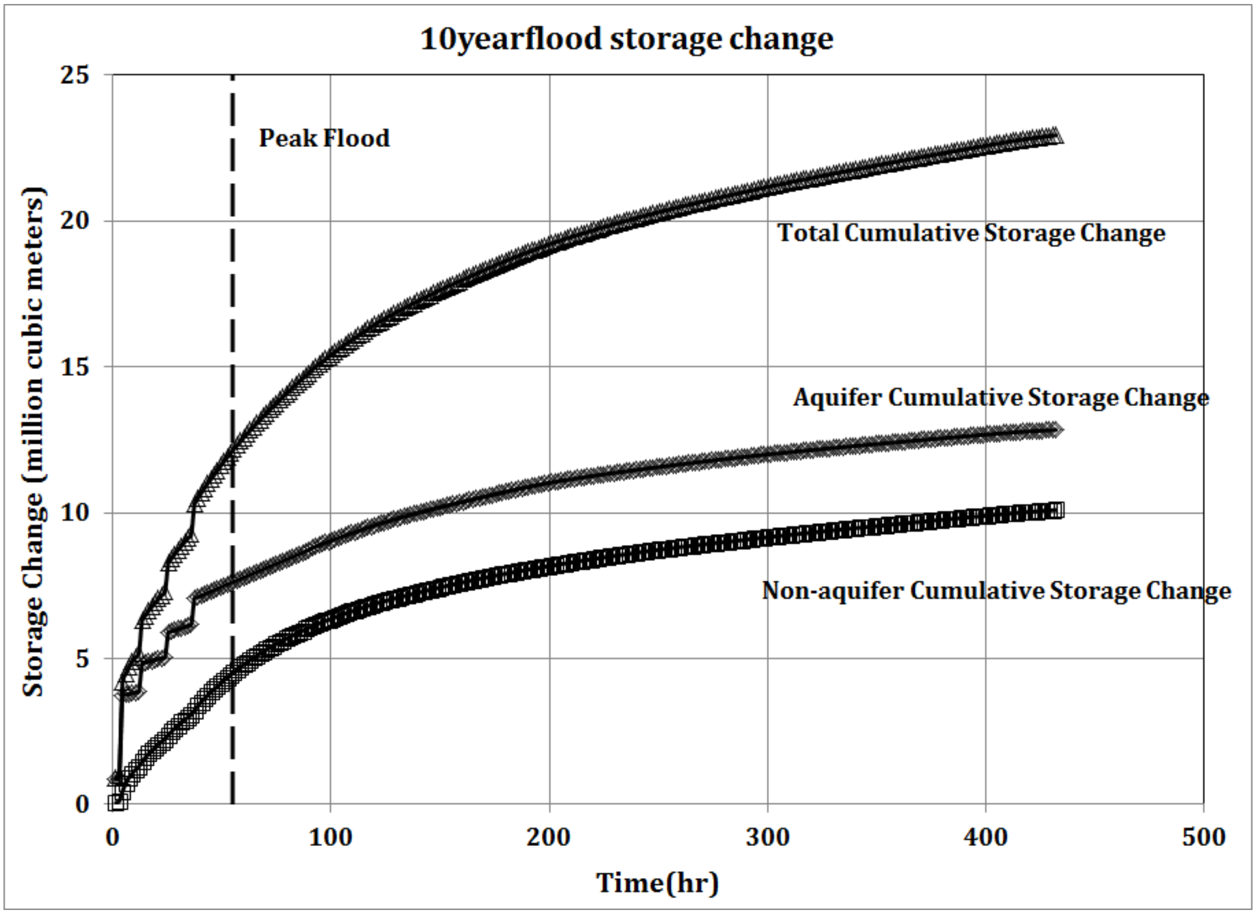


Figure 4.7B Subsurface cumulative storage change for 10 year floodplain inundation.

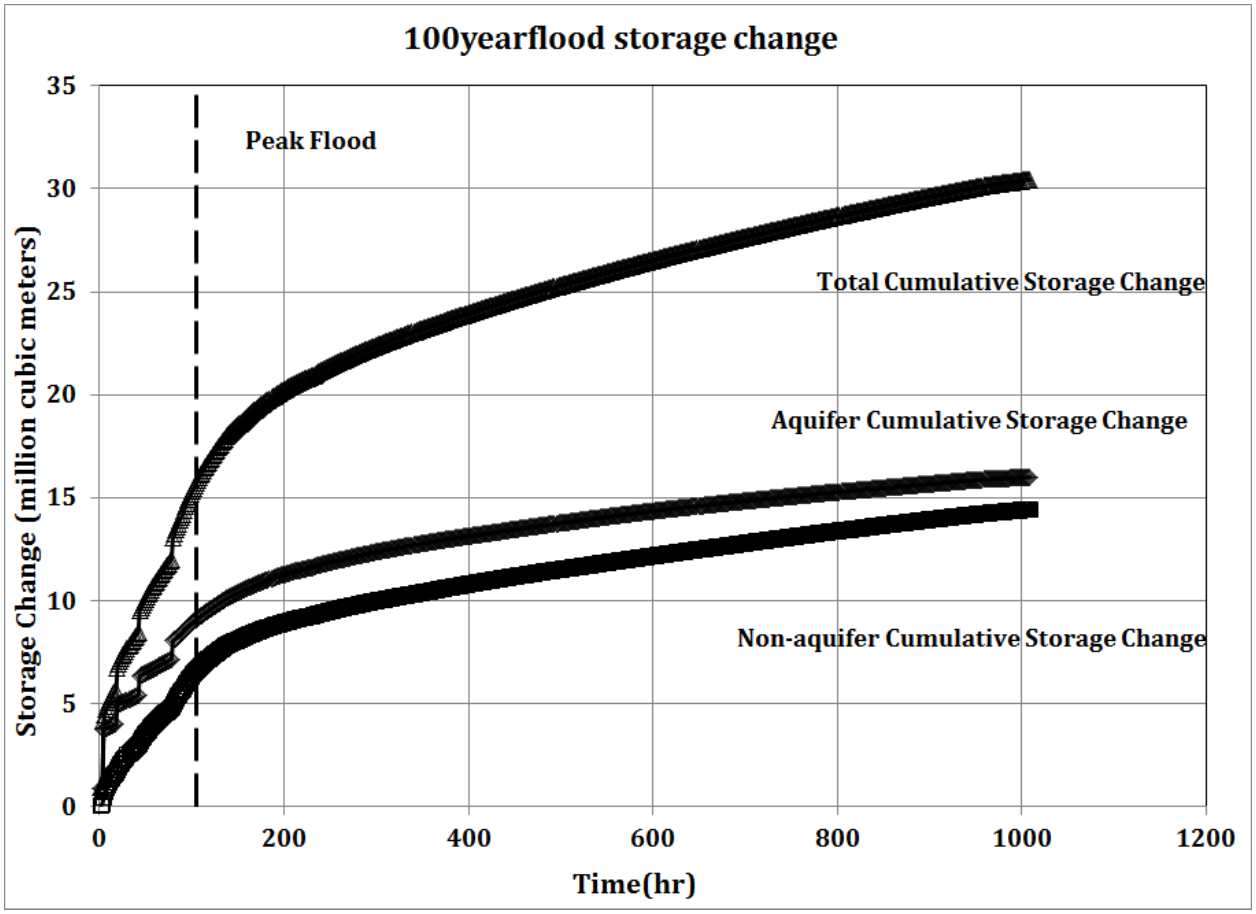


Figure 4.7C Subsurface cumulative storage change for 100 year floodplain inundation.

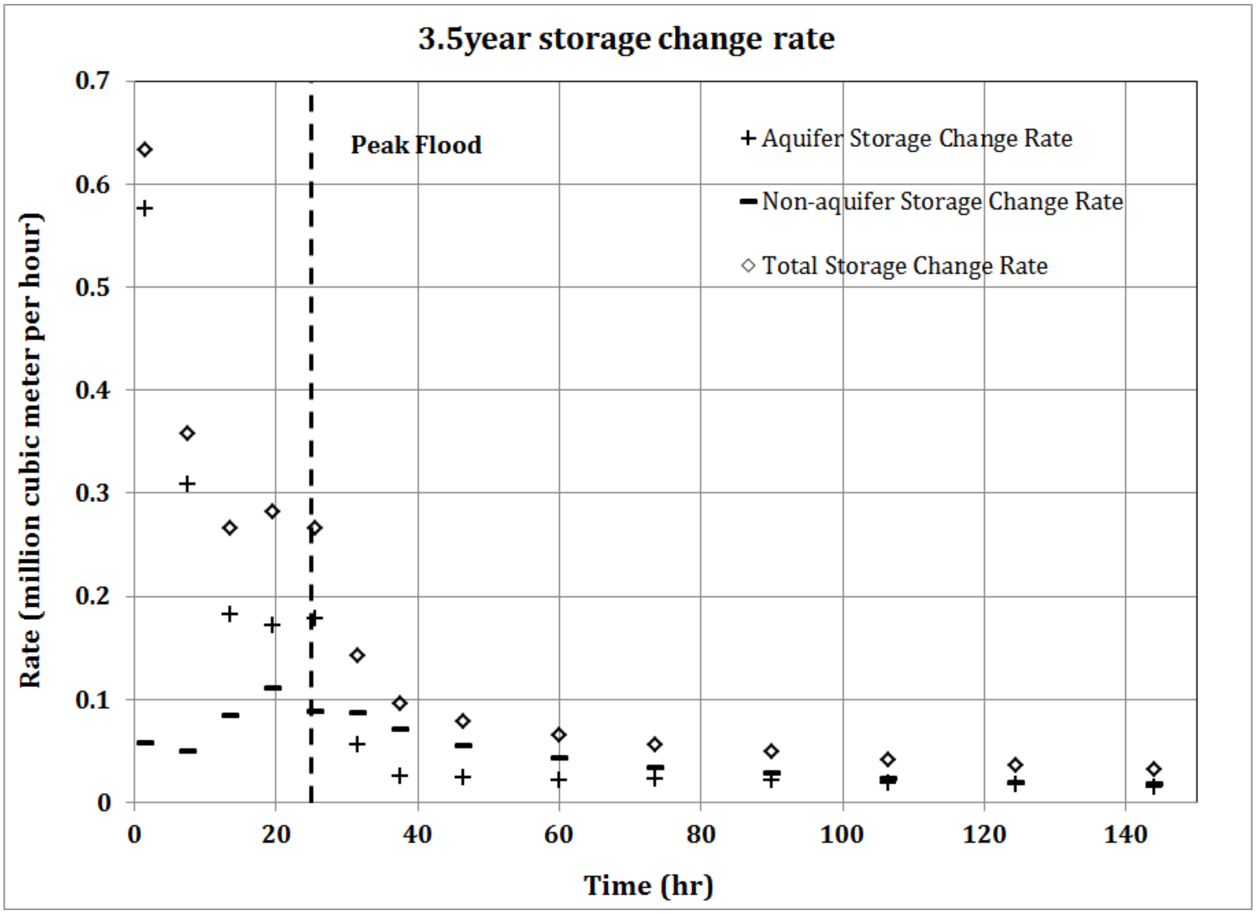


Figure 4.8 Subsurface storage change rate for 3.5 year floodplain inundation.

In summary, storage change in aquifer materials is larger than the storage change in non-aquifer materials for all three flood scenarios. Storage change is quicker before flood peak than after food peak in both aquifer and non-aquifer materials. However, a lot more storage change in aquifer materials happens before flood peak, while non-aquifer materials behave the opposite (Table 4.4). The total subsurface storage change increases with less frequent flood event (Table 4.3).

Table 4.3 Total subsurface storage change for three types of flood events (1 million cubic meters= 0.81 thousand acre feet).

Flood Type	Total storage change (million cubic meter)	Total storage change in aquifer(million cubic meter)	Total storage change in non-aquifer(million cubic meter)
3.5 year	14.53	8.47	6.06

flood			
10 year flood	22.93	12.83	10.09
100 year flood	30.44	15.98	14.45

Table 4.4 Cumulative storage change in aquifer and non-aquifer sediments before and after flood peak (1 million cubic meters= 0.81 thousand acre feet).

		BeforeFlood Peak cumulative change (million cubic meters)	**AfterFloodPeak cumulative change (million cubic meters)**	**BeforeFloodPeak averaged change rate (million cubic meters per hr)**	**After Flood Peak averaged change rate (million cubic meters per hr)**
3.5 year flood	Aquifer	5.90	2.57	0.236	0.022
	Non-Aquifer	2.07	3.98	0.083	0.033
10 year flood	Aquifer	7.63	5.20	0.138	0.0144
	Non-Aquifer	4.53	5.55	0.082	0.0145
100 year flood	Aquifer	9.07	6.91	0.086	0.007
	Non-Aquifer	6.65	7.80	0.063	0.008

To further explore the flow dynamics between aquifers and surrounding non-aquifers, an extended period of simulation for 3.5 year single flood event was performed. Assuming that the extended simulation starts right after the 144 hour floodplain inundation period, additional 6 months (4320 hours) period was simulated (scenario 4). In the extended running period, it is assumed that there is no surface inundation anymore, thus no flux exchange between subsurface and surface. The mass exchange will only happen between aquifer sediments and surrounding sediments.

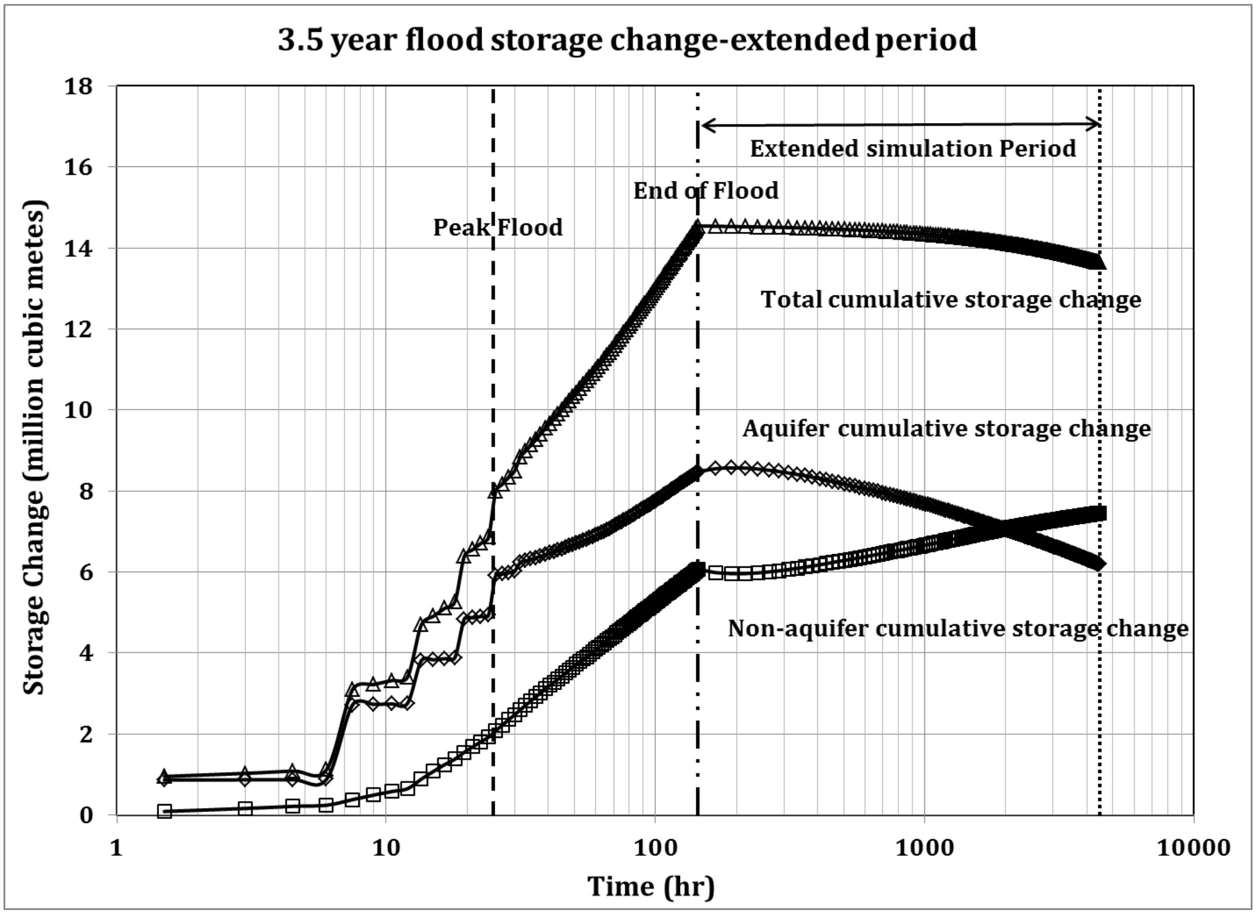


Figure 4.9 Subsurface cumulative storage change for 3.5 year floodplain inundation with extended simulation period.

Figure 4.9 is showing the subsurface cumulative storage change in both aquifer sediments and non-aquifer sediments with extended running period for 3.5 year flood event. It clearly shows that water started soaking into the low permeable non-aquifers from the aquifer materials. The aquifer materials are fully saturated after flood event due to its high hydraulic conductivity and small storage capacity, while the non-aquifer materials are still unsaturated because of its low hydraulic conductivity and large storage capacity. The difference in saturation and pressure creates hydraulic gradient toward the non-aquifers materials, which pushes water that initially in aquifers flow into non-aquifers and redistributes water horizontally. Therefore, the cumulative storage change in aquifer sediments drops while storage change in non-aquifers goes up. Even at the end of the extended period, there is still a strong increasing trend of storage change in non-aquifer sediments, which indicates the exchange process is still ongoing after 6 months. Due to the low hydraulic conductivity of non-aquifer materials, the exchange process may last for years to decades before reaching steady state (Sager 2012). The drop of total storage change seen in Figure 4.9 is due to the mass exchange between floodplain area and surrounding areas (in this study, the storage change is calculated only for floodplain and its vicinity).

For 3.5 year single flood event, total groundwater recharge from floodplain inundation is on the order of 15 million cubic meters (12.15 thousand acre feet). For 10 year single flood event, total groundwater recharge is on the order of 24 million cubic meters (19.44 thousand acre feet). For 100 year single flood event, total groundwater recharge is on the order of 31 million cubic meters (25.11 thousand acre feet) (Table 4.3). Previous studies on Cosumnes river basin show that there are 200-300 million cubic meters (162-243 thousand acre feet) of groundwater deficit annually (Fleckenstein 2004). The annual deficit in groundwater cannot be compensated by recharge from floodplain inundation as indicated by result above. However, this study is constrained only to the lower Cosumnes River floodplain, which is about 10 square miles (26 square kilometers) due to available data. It is also restricted to single flood event. The total amount of groundwater recharge can be a lot larger than the calculated value above, because of larger inundation area along Cosumnes River and multiple flooding events during winter. However, the exact amount of recharge during winter and the feasibility of offsetting groundwater deficit by floodplain recharge require long term and larger scale flood modeling. That is, modeling the entire winter season including floodplains along Cosumnes River course and floodplains at the adjunction of Cosumnes River and Delta.

Sager (2012) conducted a modeling study of artificial recharge at Cosumnes River basin and concluded that by expanding flooding area or extending flooding time, it may be possible to overcome the annual groundwater deficit. From a long term point of view, the total subsurface storage change is dominated by the storage change in non-aquifer materials. But this is a very slow process, as pointed out before. Therefore, long term, multi-year simulation may be required to examine the benefit of floodplain inundation on regional scale water budget. As one would expect, inter-annual flood frequency, magnitude, the inundation area and inundation time will be the most important factor determining the amount of recharge.

4.2.2 Surface heterogeneity and surface water groundwater interaction

The recharge across each of the four types of sediments on land surface is calculated for the entire inundation period for the three types of flood events. Table 4.5A, B and C show that the aquifer sediments (gravel and sand) outcrop accounts for roughly 26—27 percent of the surface inundated area but the recharge is about 46—54 percent of the total recharge for the entire simulation period. On the contrary, the surface area fraction of mud is more than half of the entire inundated area. The recharge across it, however, only accounts for 2—7 percent of the total recharge for the entire simulation period. The recharge across muddy sand, surprisingly, accounts for 42—46 percent of the total recharge, even though its surface area fraction is only 20 percent of the entire inundated area, which is less than the area fraction of aquifer sediments.

The limiting factor for the very low recharge across mud is, obviously, its very low hydraulic conductivity (0.0017 m/day). It forms flow barrier and obstacles downward flow. In contrast, the high hydraulic conductivity of gravel (67.52m/day) and sand (42.74m/day) creates preferential flow pathways to subsurface, thus contributes to half of the total recharge, though their combined surface area fraction is only a quarter of the inundated area. But the preferential path ways are of a small volumetric fraction and filled up quickly at early inundation time due to its high hydraulic conductivity. This reduces the hydraulic gradient between surface inundation water and near surface groundwater, which ultimately limits recharge across it. The muddy sand has an intermediate hydraulic conductivity (0.2m/day), therefor it does not form flow barrier as mud does, and it does not form preferential flow pathway as gravel and sand do. Recharge is slow in muddy sand comparing to recharge across gravel and sand because of its lower hydraulic conductivity. However, the intermediate surface area and intermediate storage capacity favors the long continuing recharge, which eventually leads to large recharge seen in the modeling results. The results here highlight the important role of intermediate permeability sediments in controlling the recharging process, through a lot of attention has been on the role of preferential path ways and flow barriers (perched aquifers).

Table 4.5A Surface recharge through each of the four types of hydro-facies -3.5yearflood.

3.5 year flood event surface recharge				
	Gravel	Sand	Muddy Sand	Mud
Cell Count	8298	6156	11808	28870
Surface area fraction	15.25%	11.32%	20.37%	53.07%
Flux(m^3)	5485145	2410460	6257510	418619
Flux fraction	37.64%	16.54%	42.94%	2.87%

Table 4.5B Surface recharge through each of the four types of hydro-facies -10yearflood.

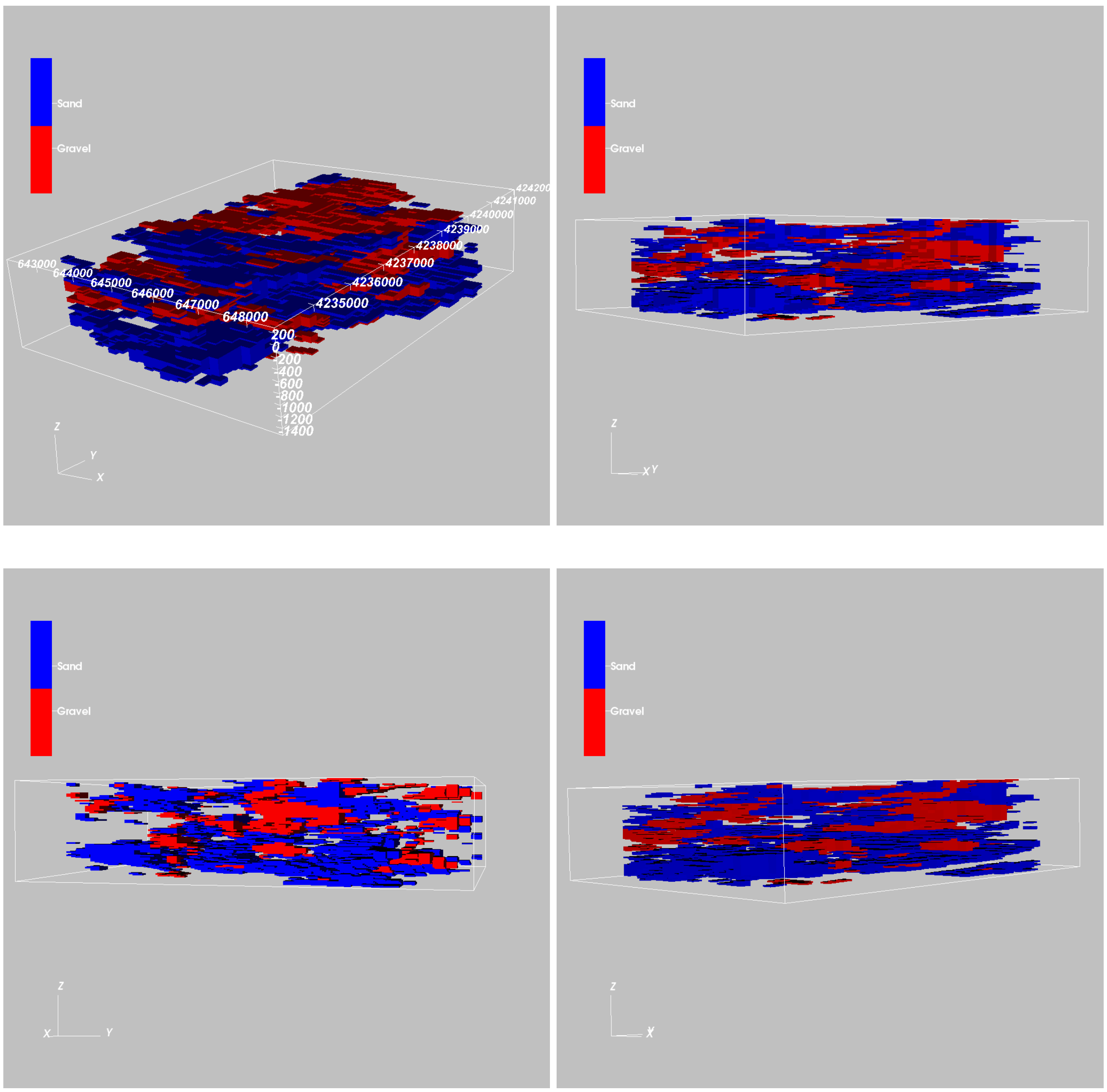
10 year flood event surface recharge				
	Gravel	Sand	Muddy Sand	Mud
Cell Count	32364	19882	41362	100424
Surface area fraction	16.68%	10.25%	21.31%	51.75%
Flux(m^3)	8109857	3259664	10715291	1177911
Flux fraction	34.86%	14.01%	46.06%	5.06%

Table 4.5C Surface recharge through each of the four types of hydro-facies -100yearflood.

100 year flood event surface recharge				
	Gravel	Sand	Muddy Sand	Mud
Cell Count	80824	47178	99298	241160
Surface area fraction	17.25%	10.07%	21.19%	51.48%
Flux(m^3)	10267891	4304400	14751956	2288248
Flux fraction	32.48%	13.62%	46.31%	7.18%

4.2.3 Subsurface connectivity and surface water groundwater interaction

Multi-aquifer systems, such as the Cosumnes River alluvial fan system studied in this modeling effort, usually show great heterogeneity. The coarse-grained sediments in this system tend to be interconnected in 3D, even though the 2D cross sections on either vertical or horizontal direction would suggest a lack of interconnection. The geometry of the connected aquifers system is very complex, as can be seen in the Cosumnes River floodplain subsurface geological structure (Figure 4.10). Two local scale features that are crucial on controlling groundwater recharge should receive special attention: the preferential pathways that connect land surface to subsurface locally, and the local vertically separated aquifer that are overlain by low-hydraulic conductivity, non-aquifer sediments. For a short term event, such as the floodplain inundation event studied in this study, the local scale heterogeneous feature can play an important role.



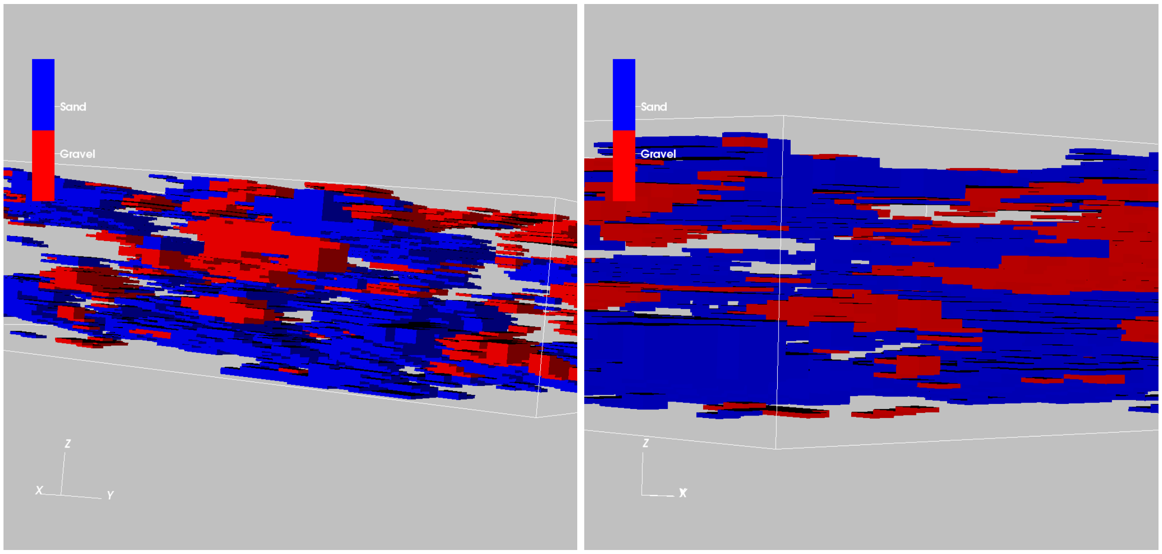
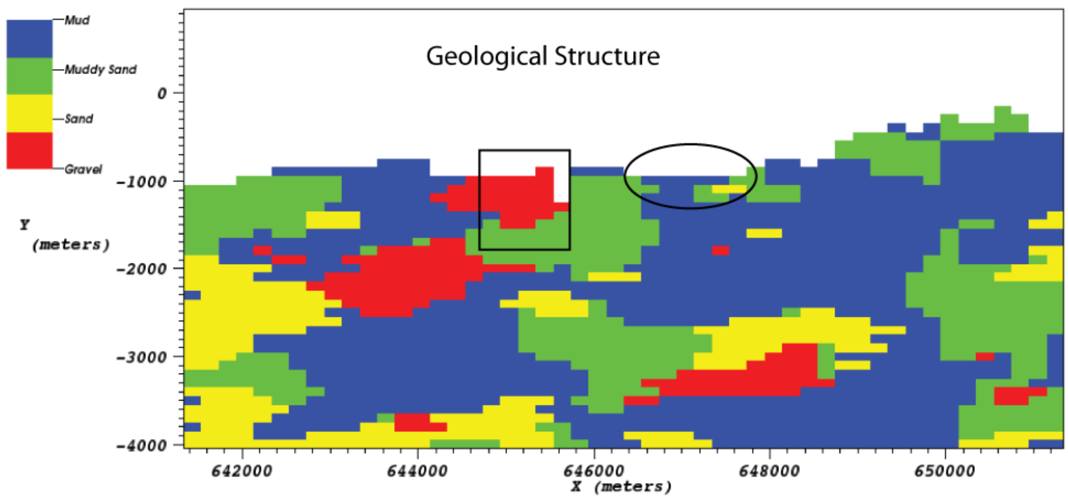


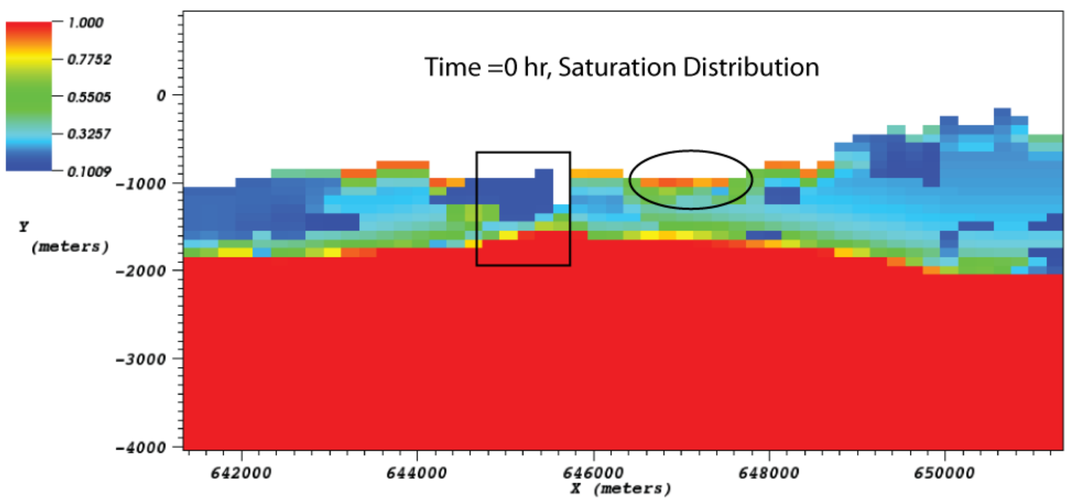
Figure 4.10 Subsurface connected aquifer system beneath lower Cosumnes River floodplain (coordinates are in WGS84-UTM10N coordinate system, vertical x30).

Figures 4.11 show the geological structure and saturation profile evolving with time at a vertical cross section of floodplain in 3.5 year floodplain inundation simulation and 3.5 year extended simulation. The square area and the circle area are two main focus areas. The geologic structure of the square area is: a large chunk of gravel that forms preferential pathway, connecting surface to subsurface, but overlies mud and muddy sand without connecting to other aquifers vertically. This is a favorable condition for perched aquifer. The geological structure of the circle area is quite simple that low permeable mud dominates. This forms flow barriers.

At time 7.5 hours, a portion of the gravel in the square has been filled up vertically and created local hydraulic connectivity between surface water and local groundwater. At time 19.5 hours, most of the gravel in the square has reached fully saturated stage. Meanwhile, in the circle, little changes in saturation profile were observed. Looking at the saturation profile in square at time 30 hours and 144 hours, it is obvious that a localized perched aquifer is formed, and the saturation front does not move downward due to the barrier effect of the low hydraulic conductivity mud that underlies the gravel. In the circle, however, the saturated front is still constrained to the near surface and did not move downward significantly.

Note the 3.5 year floodplain inundation simulation was run for only 144 hours originally. Another 6 months extended simulation was conducted with no inundation on surface. The purpose of the extended simulation is to see water redistribution phenomenon between aquifer and non-aquifers. The saturation profile at time 1104 hours shows water redistribute between aquifer and non-aquifers, that water in aquifers soaks into non-aquifers horizontally. This, however, is a very slow process. Saturation profile at 4464 hours shows only slight difference comparing to saturation profile at 1104 hours.


Geological Structure
Mud
Muddy Sand
Sand
Gravel
Y (meters)
X (meters)
0
-1000
-2000
-3000
-4000
642000
644000
646000
648000
650000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0.5505
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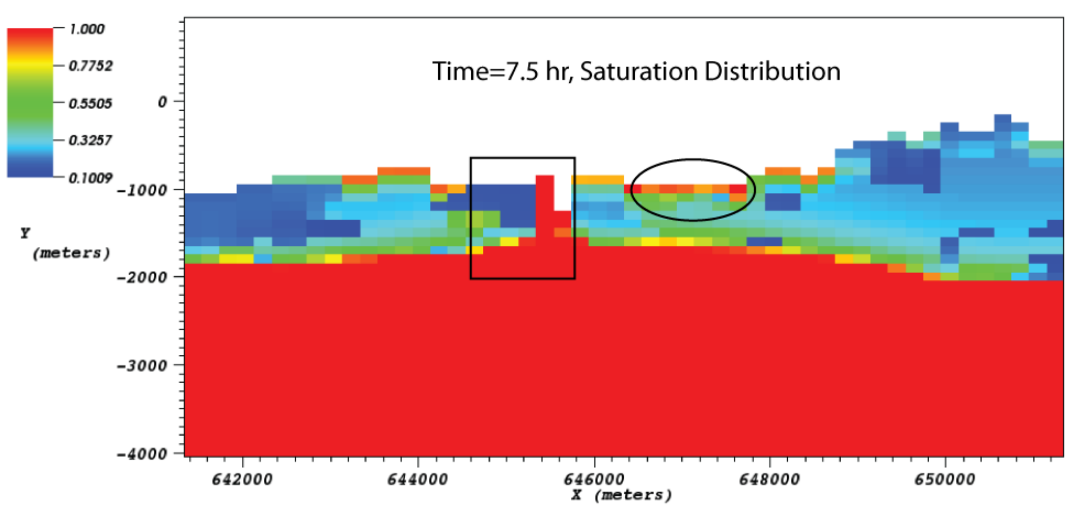
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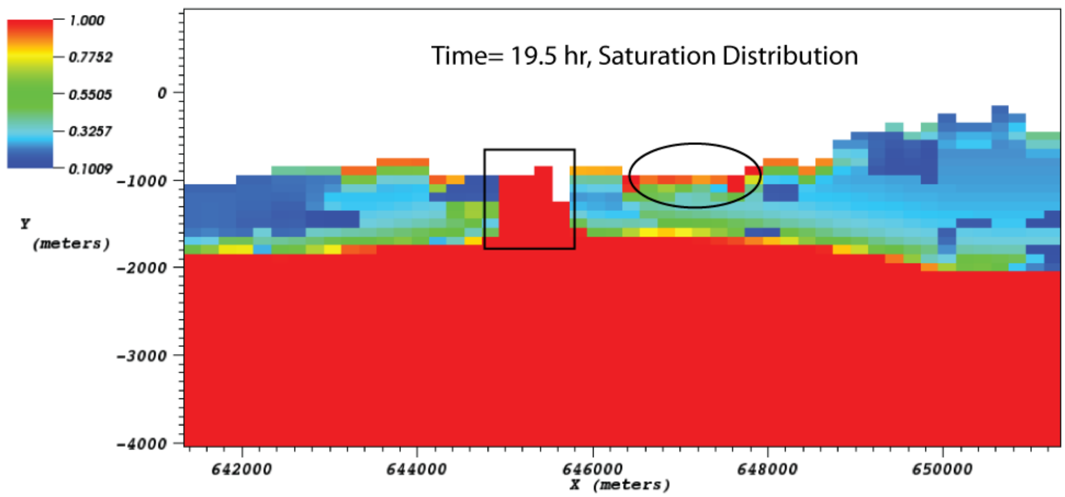
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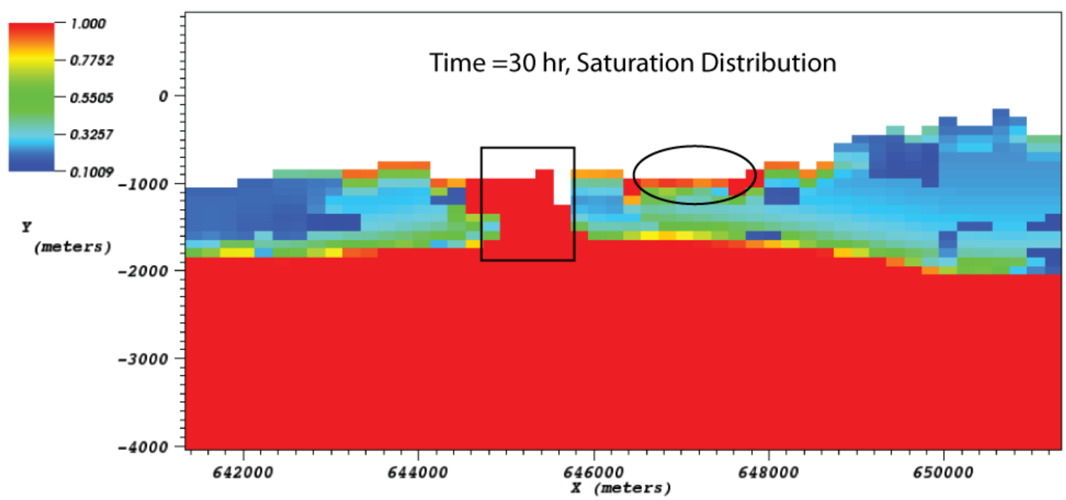
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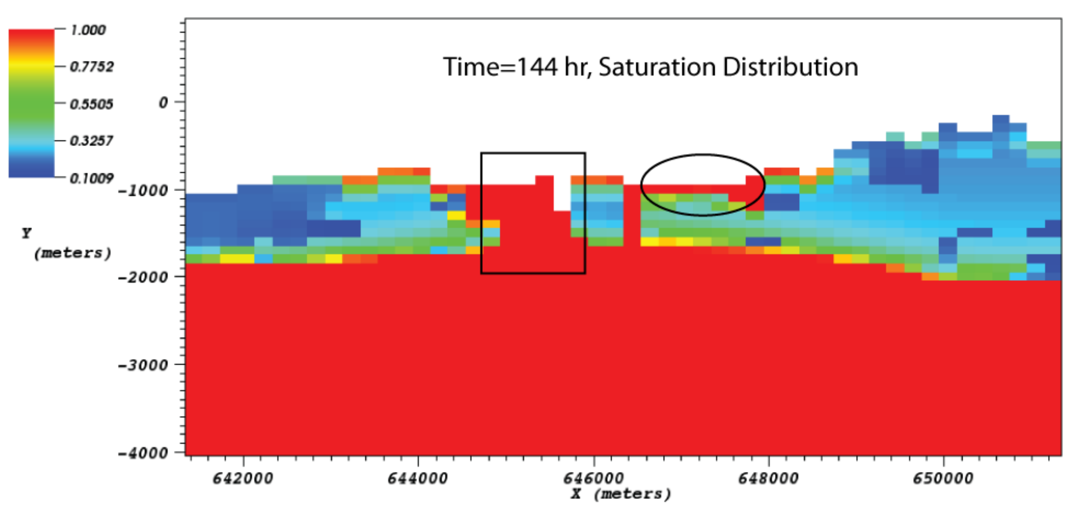
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-3000
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644000
646000
648000
650000
X (meters)

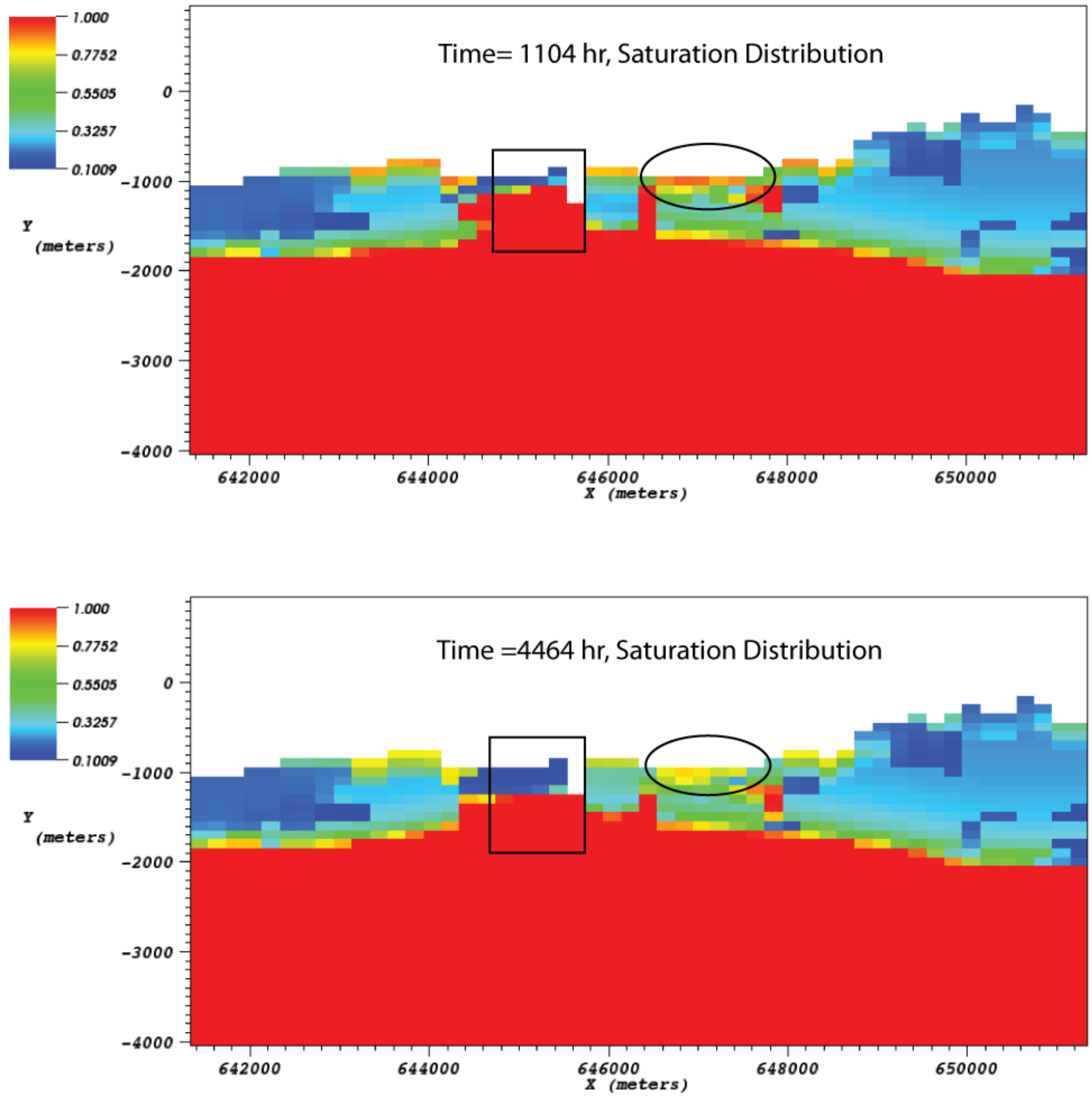

Figure 4.11 Subsurface geologic structure and saturation profile at a cross section for 3.5 year floodplain inundation event (vertical 100x).

Chapter 5 Summary and conclusions

The groundwater level in the vicinity of Cosumnes River has declined due to historical groundwater pumping, resulting in large areas of depressed groundwater levels both north and south of the river. The current groundwater level beneath Cosumnes River is far below the riverbed elevation and creates thick vadose zone. Both the cones of depression in the vicinity of Cosumnes River and the thick vadoze zone beneath the stream can potentially serve as natural groundwater reservoirs due to its large storage capacity. This is particularly important considering the storage capacity of the surface reservoirs is limited and there is very low possibility that new reservoir will be built in the future. Climate change effect makes the importance of natural subsurface reservoir even more profound. Historical observed climate data has clearly shown California is experiencing climate change on many aspects (Hamlet *et al.* 2005, Coats *et al.* 2010, Cayan *et al.* 2012). The warming trend is shifting precipitation from snow fall to rainfall, thereby reducing the snow pack in Sierra Nevada Mountains (Hamlet *et al.* 2005, Knowles *et al.* 2006, Adam *et al.* 2009). Warmer temperature also leads to earlier and shorter spring snowmelt, thereby reducing water availability of dry summer seasons (Mote *et al.* 2005, Kapnick and Hall 2010). Many other researches on California climate change suggest that the current warming trend will continue in the future, and likely bring in hotter, drier summer and wetter winters. The unbalance between most water supply period (winter) and most water demanding time (summer) with in a year and between years (dry year and wet year) demands novel strategies of capturing water at the most supply time for using at the most demanding time. Climate change effect makes this task even more challenging.

The research presented in this study provides insights on groundwater and surface water interaction, groundwater dynamics in the heterogeneous subsurface, and potential recharge benefits of winter flooding for the lower Cosumnes River floodplain. It attempts to better understand the mechanics, controlling factors of recharging process and its temporal, spatial patterns in hoping the results can guide subsurface groundwater reservoir development and conjunctive surface water and groundwater management to meet the water resource management challenges and mitigate climate change impact on California water resources.

Four floodplain inundation scenarios generated by three types of flood events were simulated, namely 3.5 year single flood event, 10 year single flood event and 100 year flood event and 3.5 year single flood event extended modeling were simulated at the Lower Cosumnes River floodplain, where floodplain recharging experiments were planned by removing the levees in favor of flooding. One of the purposes of this study is predicting of the recharging effect of the floodplain inundations experiment.

Overall, the total subsurface storage change for all of these scenarios shows similar timing and patterns. The total cumulative storage change is larger and faster at the early inundation time in both aquifer and non-aquifer sediments, particularly before the flood peak. Subsequently, the total storage change steadily goes up but at a much lower rate within the rest of flooding period. This process is highly controlled by the dynamics of flooding. Before reaching flood peak, the inundating water is expanding, covering more and more surface area associated with increasing in inundation depth. Once the flood peak is passed, the inundation area starts shrinking due to back flows to stream and inundation depth decreases accordingly. Therefore, the large and quick storage change at the early inundation time is due to the large hydraulic gradient between surface inundation water and near surface groundwater. The initially dry condition of the soil also helps drive this process. In addition, as the inundation expands, it covers more and more surface area that favors the recharge process. This also contributes to the large and quick total storage change observed in model simulation. On the contrary, as the flood peak passes, ponding depth decreases. Thus, the hydraulic gradient between surface water and near surface groundwater decreases, for near surface becomes saturated. The decreasing in hydraulic gradient plus the decreasing of inundation area leads to a lot smaller recharge and slower change in subsurface storage. However, the total amount of storage change after peak flood can be larger than that before peak flood, primarily due to longer inundation time.

The storage change in aquifer sediments (gravel and sand) and non-aquifers sediments (muddy sand and mud) show very different patterns in time and space. At early inundation time, storage change in aquifer sediments is very quick, which creates rapid jump in cumulative storage change. This is primarily due to the high hydraulic conductivity of aquifer sediments and the large hydraulic gradient between surface water and near surface groundwater. Comparatively, the storage change in non-aquifer sediments is slower and at a smaller magnitude. The low hydraulic conductivity of non-aquifer sediments is the controlling factor in this process. On the other hand, the aquifer sediments have a small volumetric fraction, thereby small storage capacity. Once it is filled up at early inundation time, very little un-used storage volume is available for the rest of flooding period. This explains the smaller storage change observed in aquifer sediments after flood peak as compared to storage change before flood peak. On the contrary, non-aquifer sediments have a large volumetric fraction, thus the storage capacity is large. Due to its low hydraulic conductivity, the storage change process is slow but steady. The larger storage change in non-aquifer materials after flood peak than that before flood peak is mainly because the flooding time after peak is longer. An interesting phenomenon observed in this modeling study is that most of the storage change in non-aquifer sediments (muddy sand and mud) actually happens in muddy sand hydro-facies, even though its volumetric proportion is a lot smaller than that of mud. After the aquifer sediments have been filled up, the intermediate permeability muddy sand becomes the secondary preferential flow path and dominates the storage change process, as was also found by Sager (2012).

The 3.5 year flood event simulation is extended for another 6 months assuming no floodplain inundation on surface for the purpose of investigating dynamics of groundwater interaction between aquifer materials and non-aquifer materials. Results of this simulation are interesting. Storage in aquifers material decrease steadily, while storage in non-aquifer increase steadily. Both are at a slow rate. This suggests water that is initially in aquifer materials soaks into non-aquifers. Recall that the aquifers are fully saturated after floodplain inundation, but the non-aquifers nearby is unsaturated. This difference creates hydraulic gradient between aquifers and non-aquifers, which forces water flow into non-aquifers from aquifers. Again, because of the low hydraulic conductivity of non-aquifers, the flow exchange process is slow. Even after 6 months, the exchanging process is still under taking, as can be seen from the increasing trend of storage change in non-aquifers and decreasing trend of storage change in aquifers at the end of simulation (Figure 4.9). The flow exchange process can last for years to decades until non-aquifers are fully saturated (Sager 2012). This time frame is important for water resource management and groundwater restoration practices.

Severe drought, like the one California is experiencing now, challenges water resource managers to come up with novel management strategies. The massive cone of depressions created by historical groundwater overdraft and thick vadose zones can potentially serve as local subsurface groundwater reservoir. Modeling results in this study help us understanding the groundwater and surface water interactions in heterogeneous vadose zone, and controlling factors of the interaction process. These are important knowledge of vadose zone management, conjunctive groundwater and surface water management and subsurface groundwater reservoir development.

Chapter 6 Model limitation and future work

In this study, the north and south boundaries of the flow model are no flux boundary condition. This can create arbitrary larger groundwater draw down responds to pumping and larger groundwater rising responds to recharging near boundaries, when pressure response reaches the boundary. Both of the north and south boundaries locate close to cone of depression and streams (American River to the north, Dry Creek to the south), where local hydraulic gradient exists. Thus, flux across both boundaries is expected. To account for these fluxes, a general head boundary condition would be more proper. River stages at American River to the north and Dry Creek to the south can be used as general head. The east boundary is specified head boundary condition, with the specified head being the average of year 1969, 1984, 2004 fall groundwater head at the east boundary. Year 1969, 1984 and 2004 are chosen because they represent dry year, wet year and normal years. However, fall groundwater head tends to be lowest within a water year, thus the specified head on the east boundary probably is lower than it should be. Instead, the average of both spring and fall groundwater data can be better.

River routing under overland flow boundary condition was used initially. But due to its long computing time, the overland flow boundary condition for streams is not implemented in later modeling. Instead, specified head boundary condition is implemented. The specified head boundary condition can lead to much larger stream recharges to groundwater. Thus, an improvement of the model could be to include overland flow boundary condition for the streams and evaluate the stream recharges to groundwater.

The initial condition applied in this study for floodplain inundation runs is obtained from a spin up run for only 1 year with 0.001m/day light rains on the top surface. As pointed out before, one year spin up run may not be long enough for the soil moisture in low permeable muddy sand and mud to reach steady state. Results from this study also pointed out that the water exchange process in non-aquifers is a very slow process. Therefore, long time (e.g. decades) spin up run can produce more realistic initial soil moisture distributions.

The groundwater and surface water interaction, groundwater recharge process is simulated and analyzed under three types of single flood events. However, all of these simulations are limited to short time frames. It will be interesting to conduct year-long simulation that includes major water supply season (winter and spring) and major water demand (summer and fall) seasons to address the winter floodplain recharge on conjunctive water resource management. It will also be interesting to conduct simulation on even larger time scale, such as multi-years to decades that includes wet, dry and normal years to investigate the long term benefits of winter floodplain recharge on water resource management and mitigating climate change impact on water resources. In addition, the role of subsurface reservoir on water resource management can also be addressed directly in long term simulating.

The current version of the model does not include land surface mode (e.g. CLM), but it is not hard to link to it. An improvement of the model set up would be including Community Land surface Model (CLM) with the current groundwater surface water interaction model, which gives the model the power of addressing land uses change, climate change impact on groundwater and surface water interaction, water resource management in a more sophisticated way.

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Appendix A

A.1 Pumping well data process

The primary source of the pumping well data is the well log data set for Transition Probability Geo-statistics (TPROGS) simulation (Merovitz 2010). The monitor wells and the shallow pumping wells (less than 100 feet deep) were excluded, because their pumping is small comparing to the other deep wells. The original pumping well location is as following (Figure A.1).

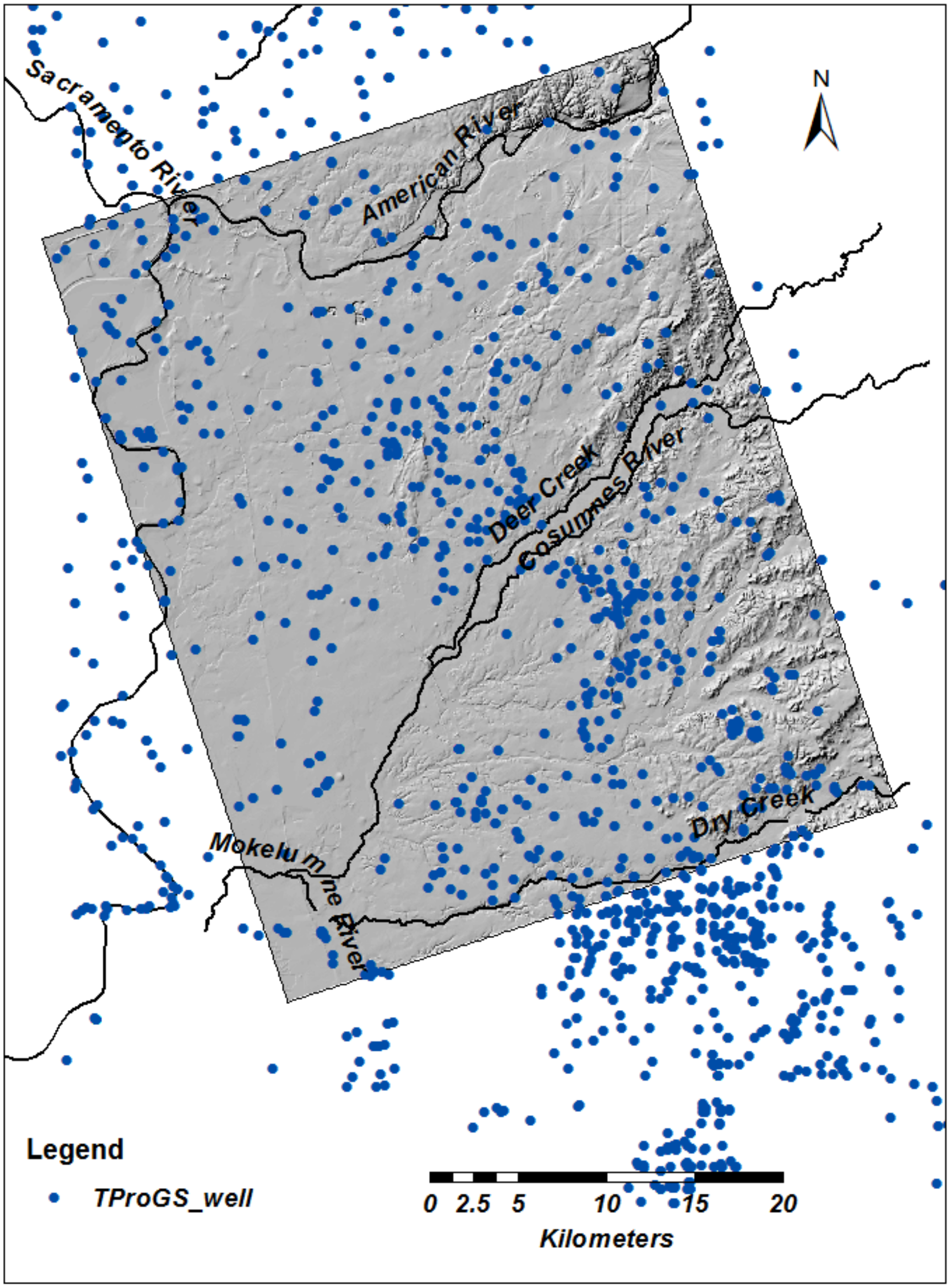


Figure A.1 TPROGS well log data wells.

Shown in Figure A1, the wells are clustered. In the model calibration process, additional pumping wells were added when necessary to allocate pumping amounts that were transferred from the prior, SACIWRM model (RMC, 2011) to the present model. It is reasonable to add additional pumping wells to the model, mainly because, 1) there are likely thousands of pumping wells in the study area. The well log data set for TPROGS simulation accounts for only a small portion of the total pumping wells (Personal Communication, Nick Newcomb, 2013), 2) the agricultural pumping estimates used in SACIWRM model are not specific to any well, but are allocated to model element regions based on crop-consumptive use calculations (reference the SACIWRM report). Final pumping wells are shown in Figure A.2.

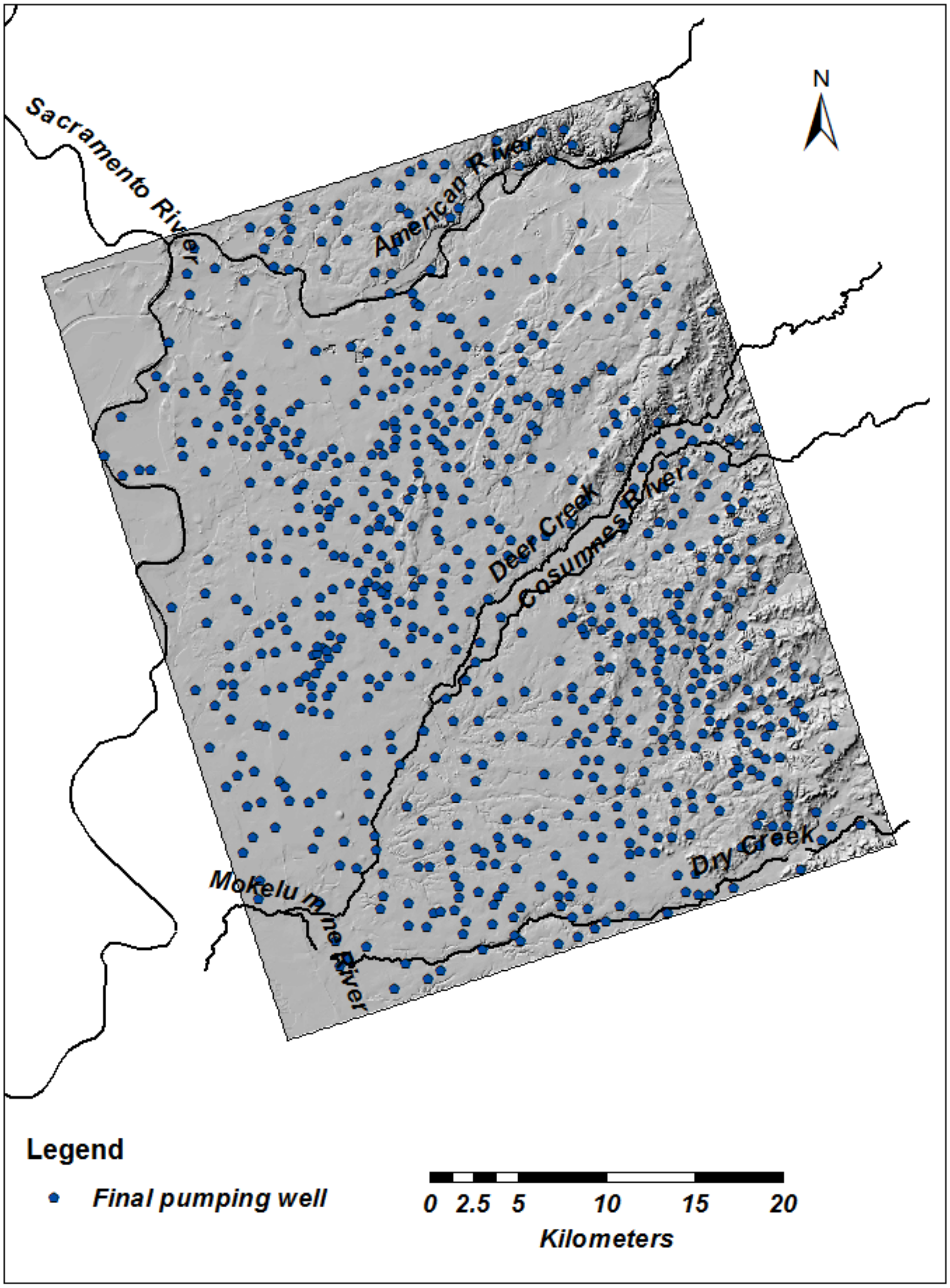


Figure A.2 Final pumping well location map.

A.2 Pumping rate data process

The primary source of the pumping rate is from SACIWRM model (RMC 2011). SACIWRM divides the study area (Sacramento County) into sub-regions based on water purveyors boundaries, land use type, political boundaries, hydrologic features and water supply features (RMC 2011). The same sub-region division was used in the ParFlow groundwater and surface water interaction model. The pumping rate for each sub-region was added up and evenly distributed to each of the pumping wells within the same sub-region initially. In addition, in ParFlow groundwater and surface water interaction model uses the same sub-region division as SACIWRM model. The sub-regions are categorized into urban region and agricultural region (Figure A.4). In the calibration process, the pumping rate for wells in the urban region is kept un-changed, while the pumping rate for wells in the agricultural region is slightly adjusted when necessary in order to achieve the calibration goals.

These adjustments in pumping rates for agricultural region are justified in this case because the pumping and recharge estimates from the SACIWRM model stem from a calibration of that model, which is based on a conventional, essentially homogeneous model. Because the current model has a very different, but more realistic geologic structure, and because the crop consumptive use method of estimating groundwater recharge is only approximate, the pumping of agricultural region is adjusted as long as the adjustment falls within the error bounds (typically less than 20-50 percent) of crop consumptive use method. The calibration is to determine whether the heterogeneous model is plausible not only in the context of available geological data, but also the water budget information as presented by SACIWRM. In other words, the purpose of the calibration in this case was not to develop a predictive model, but rather to determine whether the heterogeneous model is both plausible and consistent with the water budget, accounting for the fact that the water budget itself is approximated.

Figure A.3 shows the percentage of adjustment in groundwater pumping for agricultural regions. The maximum percentage of adjustment is less than 15 percent, which indicates the pumping applied in this model is OK.

There is no data available on the well screen depth for pumping wells, thus for each of the pumping well, multiple well screens were applied. The setup of multiple screens is by searching all continuous gravel and sand sections vertically at a given well location and assume screen locate at those section with vertical interval lager than 4 meters. Total pumping for a well is evenly distributed to each of the screens.

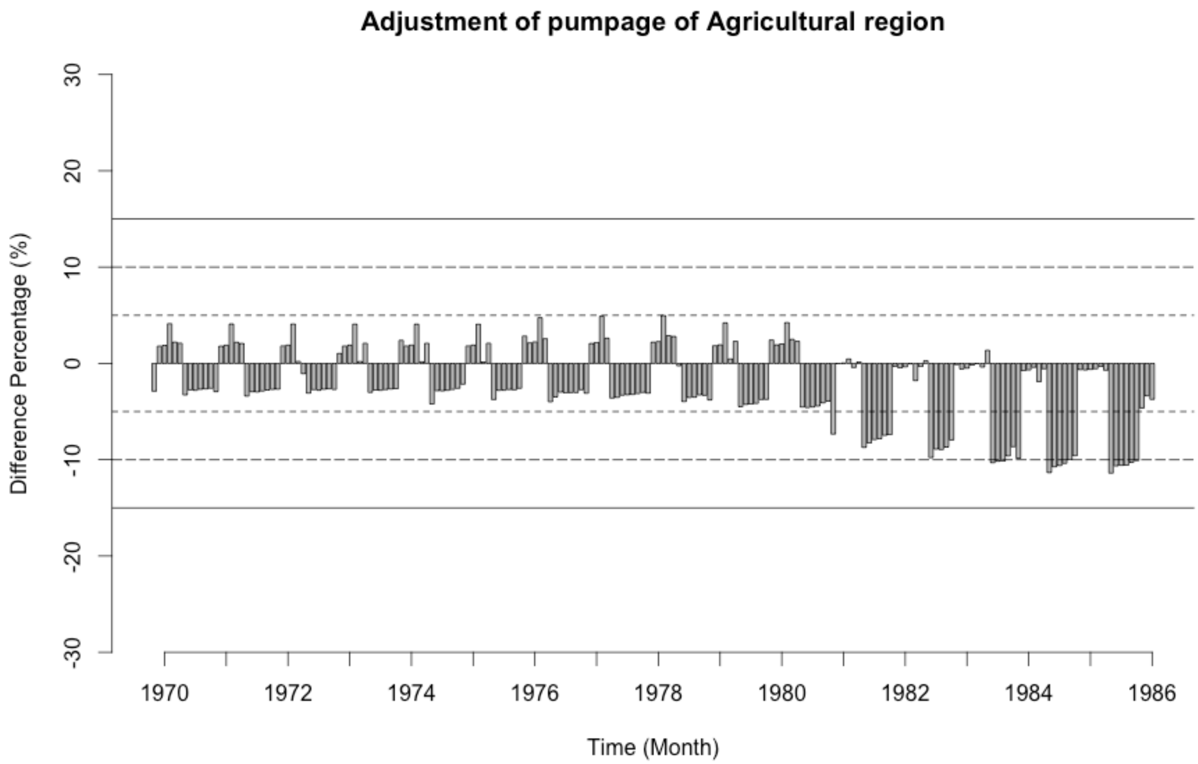


Figure A.3 Percentage of adjustment in pumping for agricultural region of the calibration period.

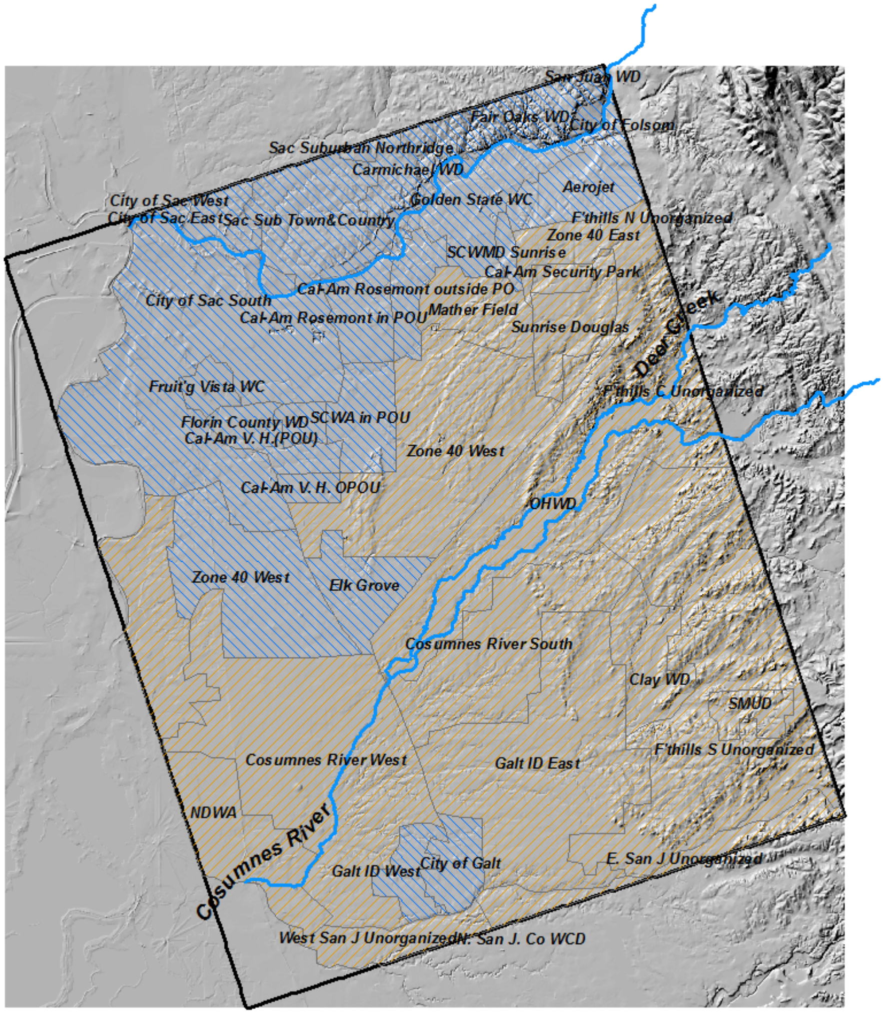


Figure A.4 Categorization of sub-regions (blue is urban area, orange is agricultural area).

A.3 Deep percolation data process

The major source of the deep percolation data is from SACIWRM model (RMC 2011). The deep percolation was mapped onto the ParFlow groundwater and surface water interaction model surface on a monthly base. As an example, 1970 August deep percolation is shown in Figure A.5.

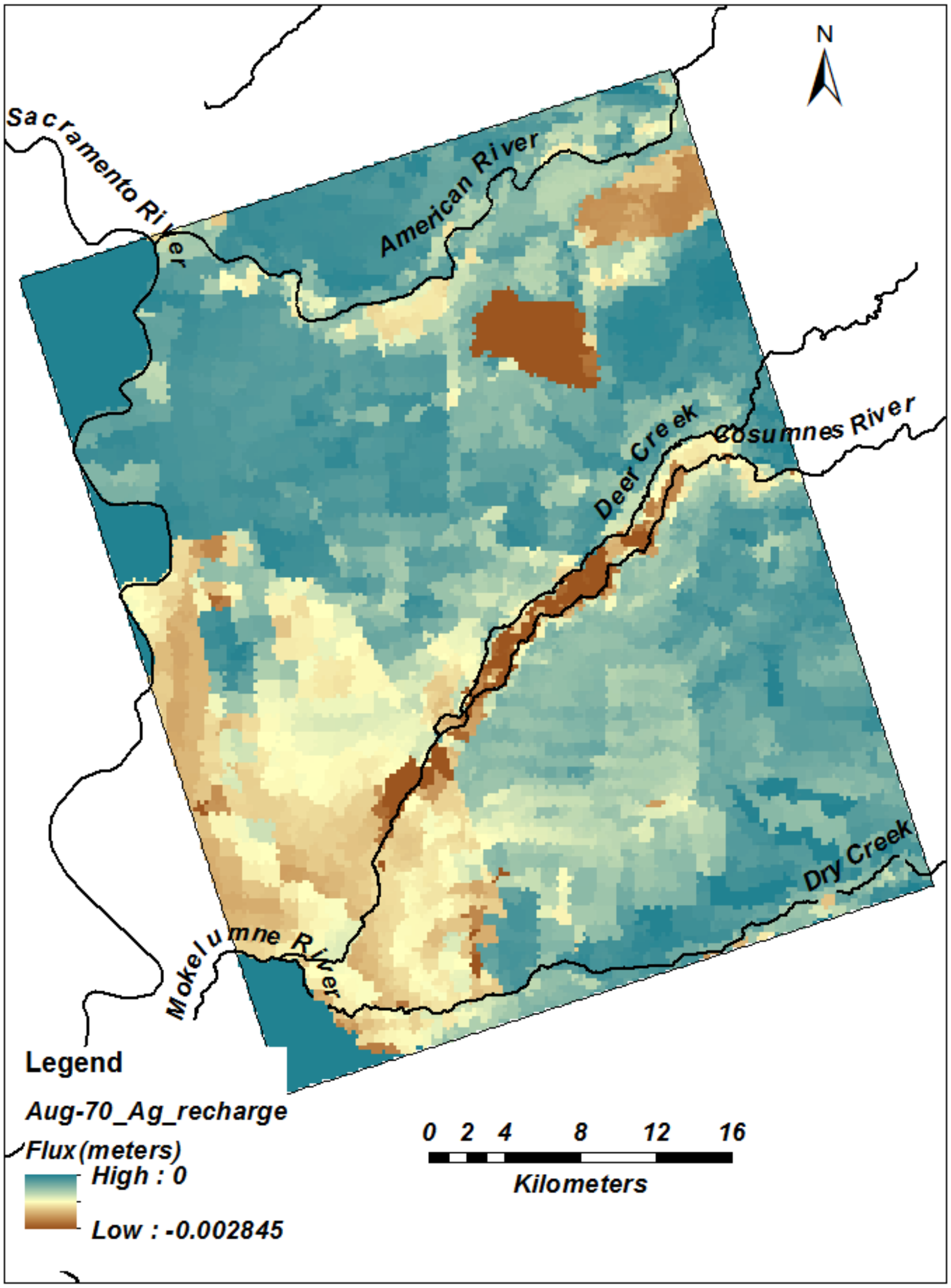

N
Sacramento River
American River
Deer Creek
Cosumnes River
Dry Creek
Mokelumne River
Legend
Aug-70_Ag_recharge
Flux(meters)
High : 0
Low : -0.002845
0 2 4 8 12 16
Kilometers

Figure A.5 Agricultural recharge for the month August-1970(negative recharge means flux into subsurface, more negative indicate more recharge).

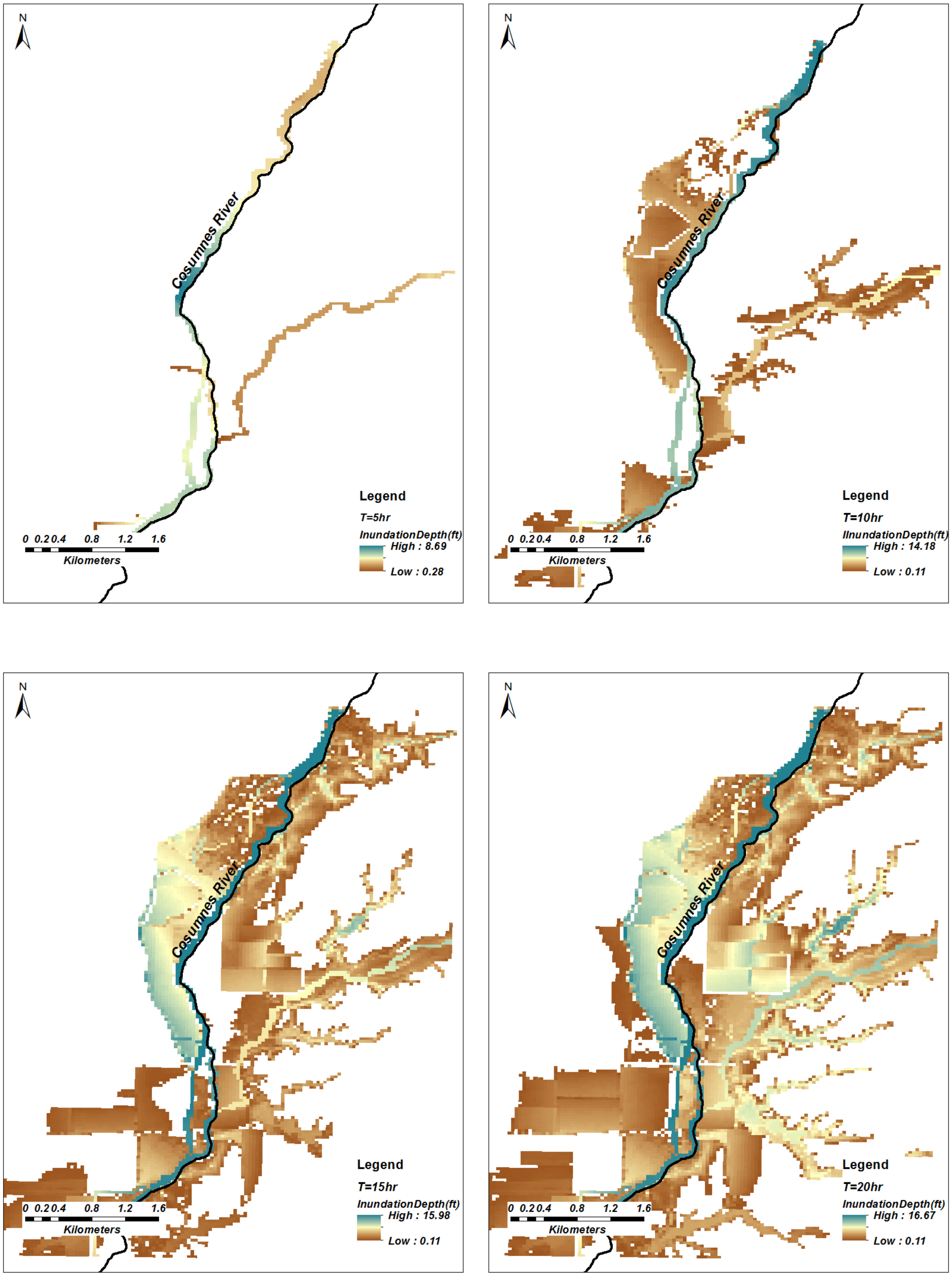
A.4 River stage data process

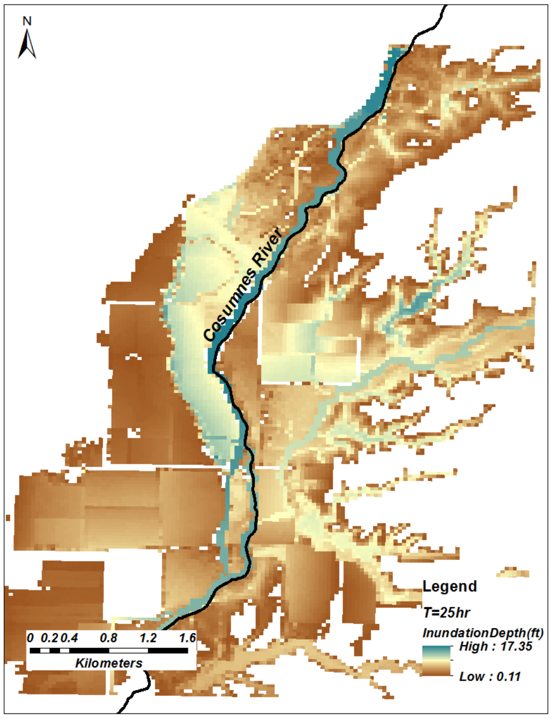
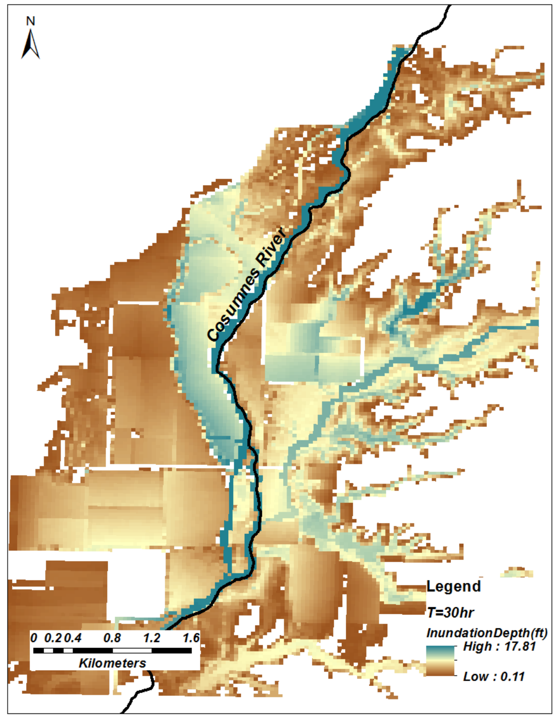
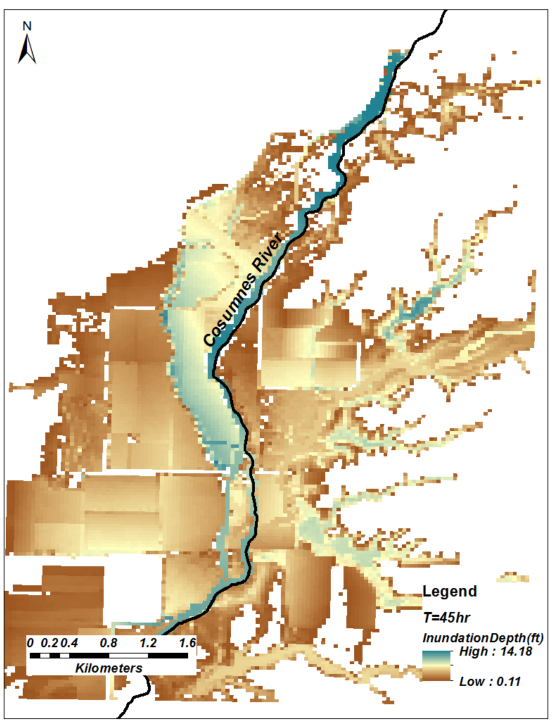
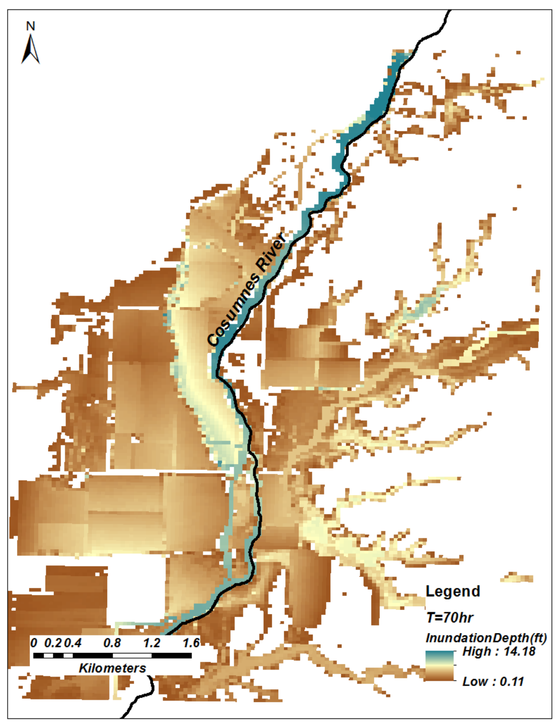
The major source of the river stage data is from SACIWRM model (RMC 2011). These streams include American River, Cosumnes River and Deer Creek. Daily river stage from SACIWRM stream nodes were interpolated and mapped to the stream course in ParFlow groundwater and surface water interaction model. Weekly averaged stream stage was used in modeling. The main purpose of including river is to capture the river seepage into groundwater rather than analyzing stream flow pattern. Thus the weekly averaged stream stage is considered adequate.

Initially, the river is applied as overland flow boundary condition, where the USGS station daily discharge is applied to upstream of rivers. However, the frequent changing of river discharge and the complexity of the model causes very long computing time. Thus, the overland flow boundary condition is not implemented for stremas. Instead, specified head boundary condition is applied.

A.5 Floodplain inundation data process

The major source of the floodplain inundation data is from RBI FLO-2D floodplain flooding simulation (RBI 2013). 3.5 year single flood event, 10 year single flood event and 100 year single flood event data were extracted. Inundation map at selected time are attached for all three types of flood events (Figure A.6, A.7, A.8). In RBI FLO-2D simulation, the 3.5 year flood event, 10 year flood event and 100 year flood event were run for 140 hours, 140 hours and 240 hours respectively. However, the floodplain flooding time frame for 10 year and 100 year flood are much longer than 140 hour and 240 hours. Booth *et al.* (2006) conducted analysis on classifying flood event in Cosumnes River basin based on the flood peak discharge. According to Booth *et al.* (2006) works, the floodplain flooding time for 10 year flood and 100 year flood event are estimated to be 432 hours (18 days) and 1008 hours (42 days), respectively.


N
Cosumnes River
Legend
T=5hr
InundationDepth(ft)
High : 8.69
Low : 0.28
0 0.2 0.4 0.8 1.2 1.6
Kilometers
N
Cosumnes River
Legend
T=10hr
IInundationDepth(ft)
High : 14.18
Low : 0.11
0 0.2 0.4 0.8 1.2 1.6
Kilometers
N
Cosumnes River
Legend
T=15hr
InundationDepth(ft)
High : 15.98
Low : 0.11
0 0.2 0.4 0.8 1.2 1.6
Kilometers
N
Cosumnes River
Legend
T=20hr
InundationDepth(ft)
High : 16.67
Low : 0.11
0 0.2 0.4 0.8 1.2 1.6
Kilometers


N
Cosumnes River
Legend
T=25hr
InundationDepth(ft)
High : 17.35
Low : 0.11
0 0.2 0.4 0.8 1.2 1.6
Kilometers

N
Cosumnes River
Legend
T=30hr
InundationDepth(ft)
High : 17.81
Low : 0.11
0 0.2 0.4 0.8 1.2 1.6
Kilometers

N
Cosumnes River
Legend
T=45hr
InundationDepth(ft)
High : 14.18
Low : 0.11
0 0.2 0.4 0.8 1.2 1.6
Kilometers

N
Cosumnes River
Legend
T=70hr
InundationDepth(ft)
High : 14.18
Low : 0.11
0 0.2 0.4 0.8 1.2 1.6
Kilometers

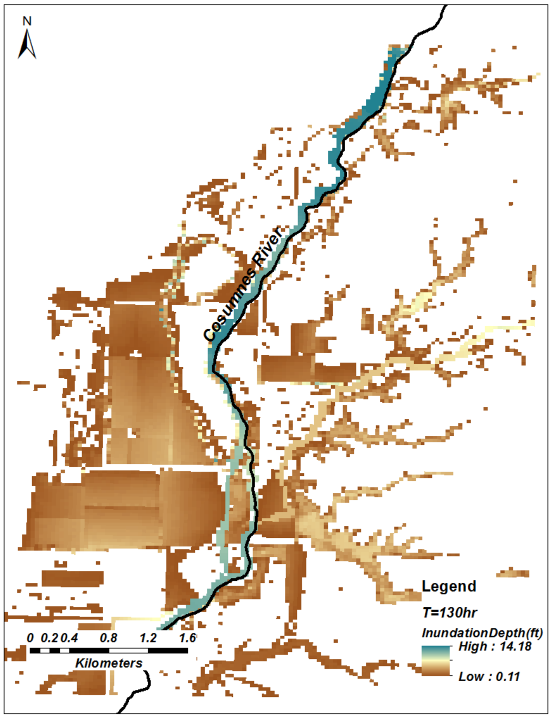
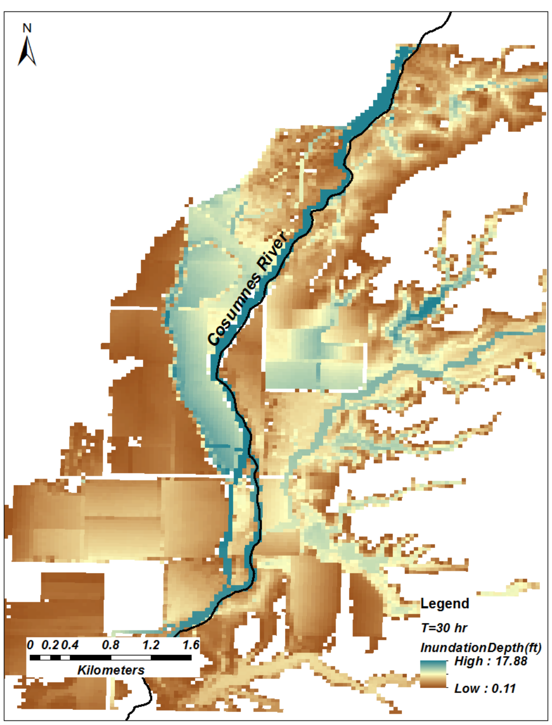
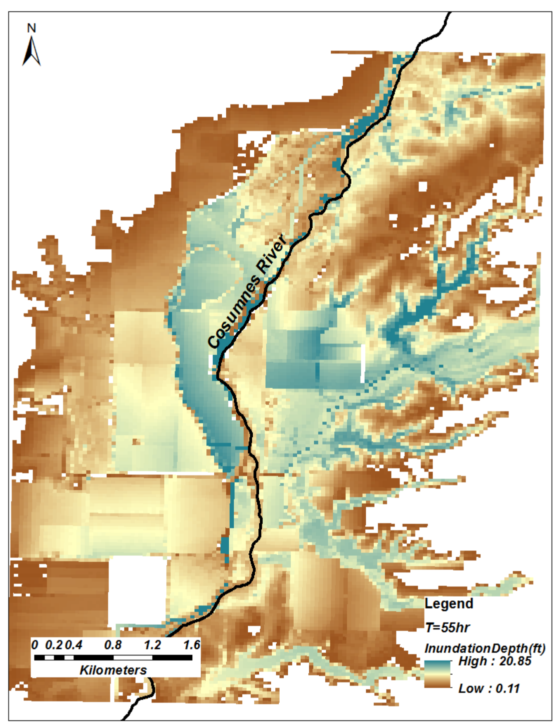


Figure A.6 Inundation map for 3.5 year flood event at time 5, 10, 15, 20, 25, 30, 45, 70 and 130 hour respectively. See Figure 3.5A for location of the above map within the model area of this study.





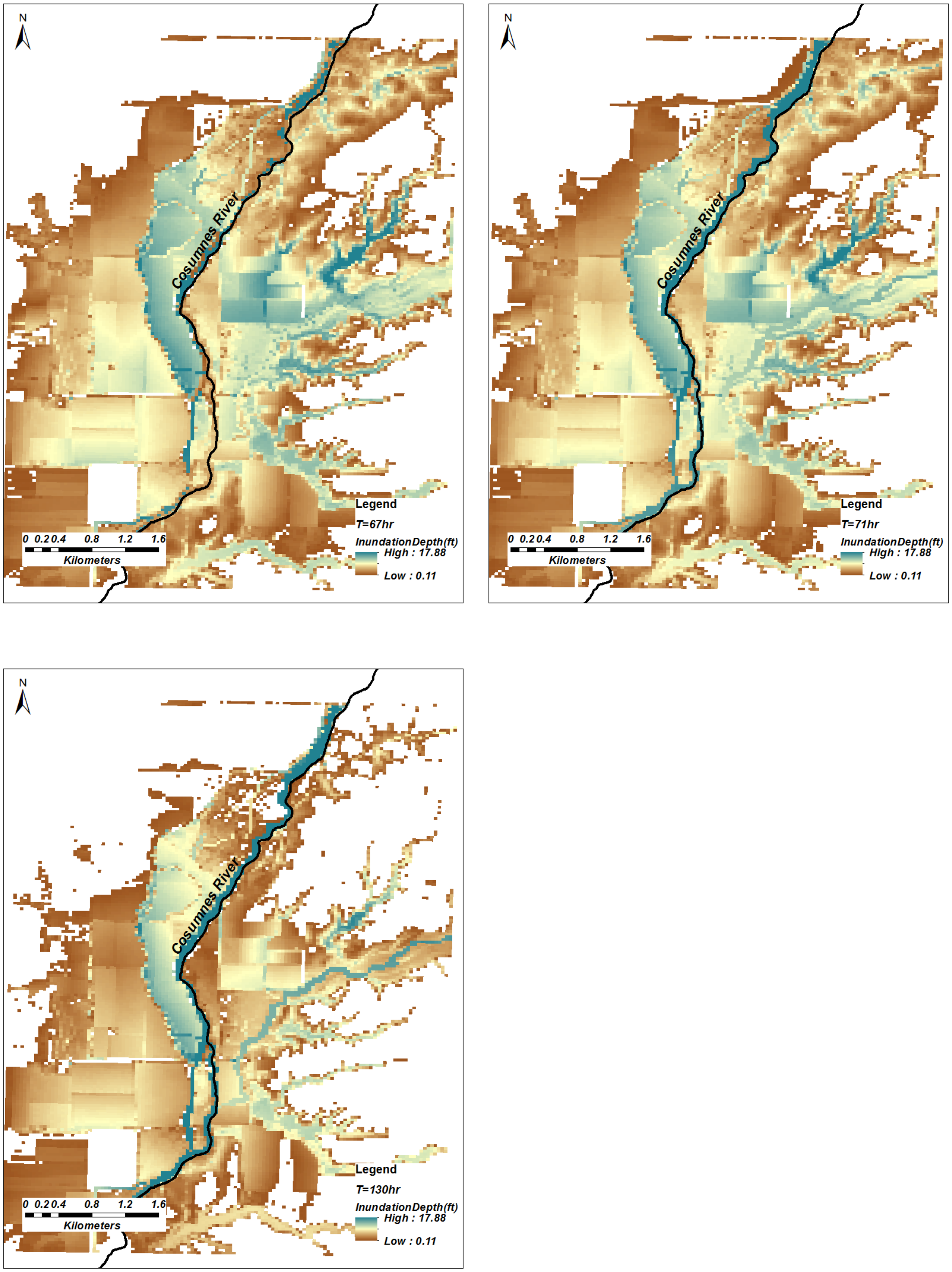
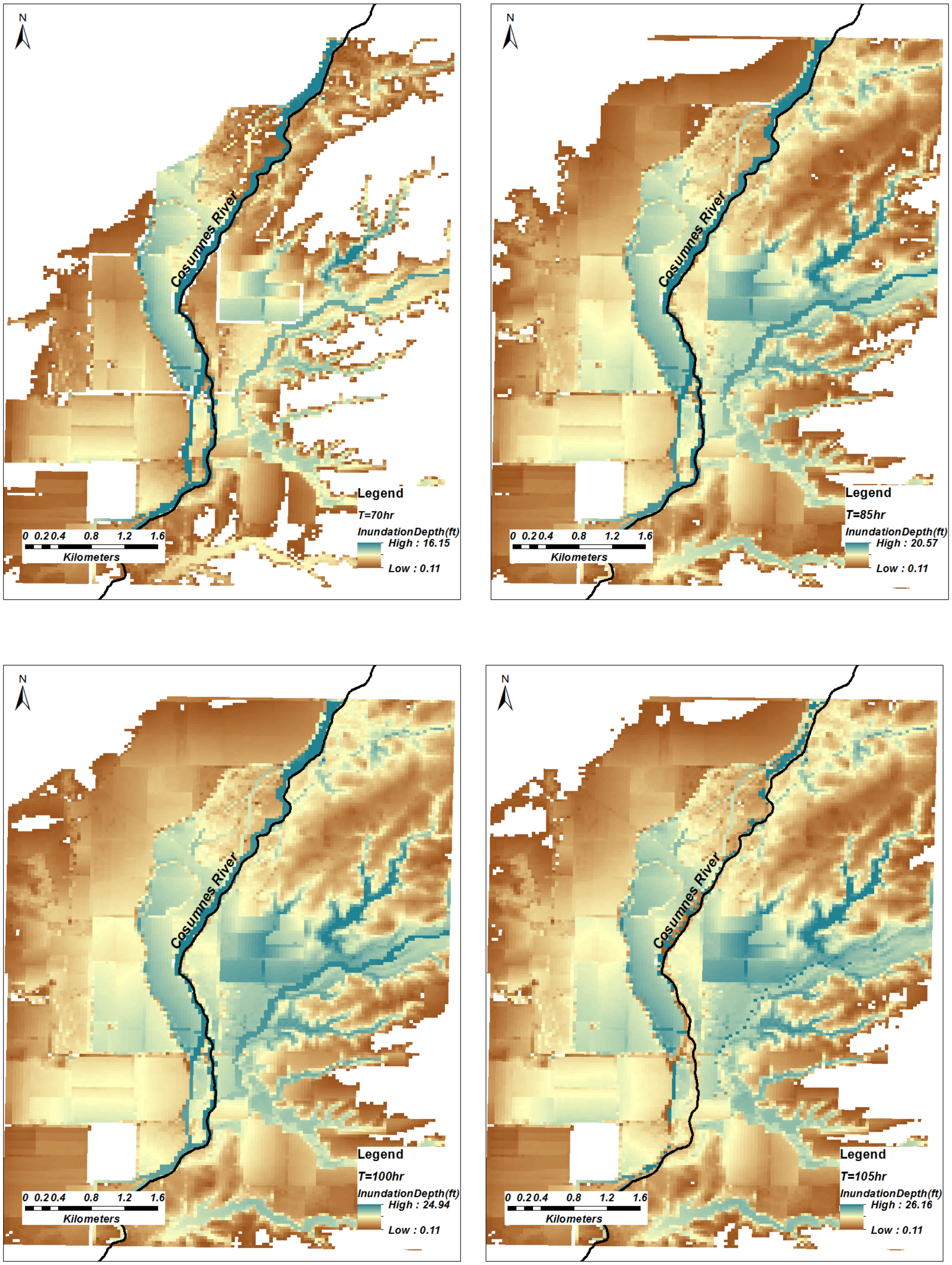


Figure A.7 Inundation map for 10 year flood event at time 30, 55, 67, 71 and 130 hour respectively.


N
Cosumnes River
Legend
T=70hr
InundationDepth(ft)
High : 16.15
Low : 0.11
0 0.2 0.4 0.8 1.2 1.6
Kilometers
N
Cosumnes River
Legend
T=85hr
InundationDepth(ft)
High : 20.57
Low : 0.11
0 0.2 0.4 0.8 1.2 1.6
Kilometers
N
Cosumnes River
Legend
T=100hr
InundationDepth(ft)
High : 24.94
Low : 0.11
0 0.2 0.4 0.8 1.2 1.6
Kilometers
N
Cosumnes River
Legend
T=105hr
InundationDepth(ft)
High : 26.16
Low : 0.11
0 0.2 0.4 0.8 1.2 1.6
Kilometers

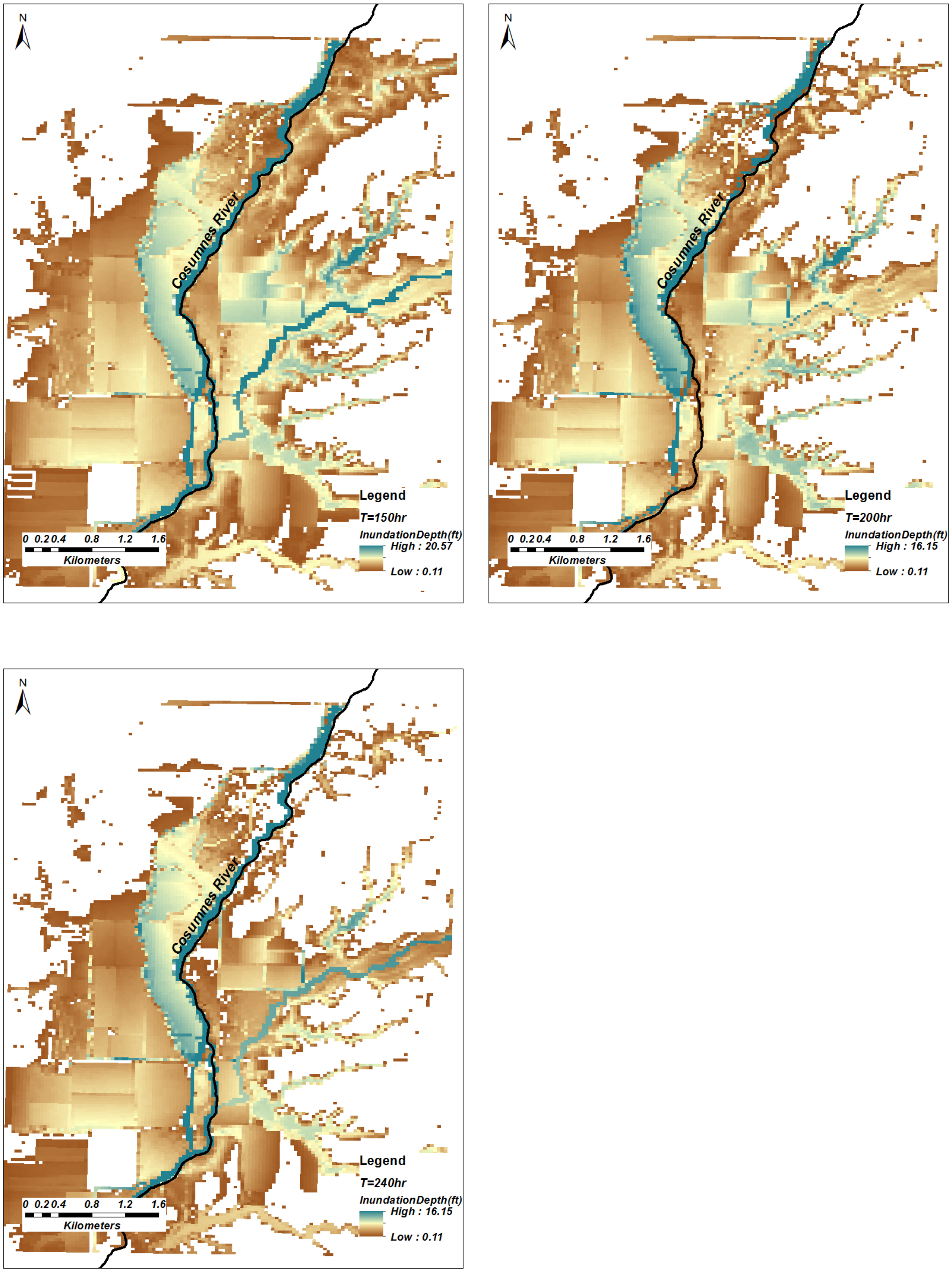


Figure A.8 Inundation map for 100 year flood event at time 70, 85, 100, 105, 150, 200 and 240 hour respectively.

Appendix B

B.1 Stream mapping and flow direction enforcement scheme

The alluvial fans are mostly flat with mild elevation variations. In numeric model, the scales of topography that could be resolved is limited due to the computational constrains (Daniels *et al.* 2011). Thus with insufficient cell resolution, the river channels cannot be resolved by simply mapping local Digital Elevation Model (DEM) on to the numerical model surface. Daniels proposed a method on mapping river course on to model surface and forcing river routing along the stream, which consists two steps: 1) retrieve and map flow line points data from the National Hydrography Dataset (NHD) to the local DEM, 2) sort the river representing points so that each point is next to its upstream and downstream neighbor, then assign flow direction accordingly (Daniels *et al.* 2011).

Many hydrological models rely on the slope magnitudes and directions to determine surface water routing direction and quantity. Therefore, the fact that DEMs used in hydrological model are often too coarse to resolve river channels makes it hard to properly set up river on to the model surface by using DEM only. A scheme of incorporating NHD flow line into DEM and map onto model surface, then assigns flow directions and slope magnitude manually for stream routing worked very well in a study on Owens Valley, CA (Daniels *et al.* 2011). The scheme in Daniels *et al.* 2011 is summarized as below.

The river representing cells on the surface of the model domain was determined by comparing the latitude and longitude of the NHD flow line points with the coordinates of model cells. The model cell that is closest to the location of river flow line point is chosen. Then these model cells are sorted and labeled from upstream to downstream. The flow direction enforcement algorithm takes in the river representing cells and determines a continuous flow path by ordering the cells along the river so that each cell is followed by another neighbor cell (cells that share a side). Eventually, the river is represented by a series of ordered cells that with a slope pointing to its downstream neighbor (Daniels *et al.* 2011). The riverbed elevation is the extracted from DEM and the slope of each cell is calculated and averaged along the river channel based on the riverbed elevation at the inlet and outlet of the river.

As the coarse resolution DEM cannot resolve river channels and the alluvial fan is mostly flat, the riverbed elevation from DEM can be wrong. The riverbed elevation is assumed to be lowered by 2 meters artificially to insure the river incises into the floodplain (Daniels *et al.* 2011). However, field observations need to be conducted to validate this assumption.

B.2 Test of river flow routing

ParFlow couples subsurface flow to surface flow via an overland flow boundary condition, with the kinematic wave equation applied at the land surface (Kollet and Maxwell 2006). The kinematic wave formulation of the overland flow routing depends only on the land surface slope and surface ponding depth of computing cell for calculating how much water moves downstream.

The goal of tests conducted in this section is to check whether the stream mapping and flow direction enforcement schemes works correctly and appropriately in ParFlow. In addition, these tests examine whether stream flow is routed smoothly along the stream course and the "water tower" problem is avoided (water cumulates at certain computing cell, resulting in very large pressure) (Daniels *et al.* 2011).

A simple test model is developed. The test model is the same size as the final groundwater surface water interaction model and has the same cell discretization. The only difference is that the test model has only 10 homogeneous layers. The subsurface is treated as fully saturated homogeneous low permeable sediments, in order to minimize execution times devoted to the groundwater flow solver. The surface of the model is assumed to be flat, but with arbitrary slope applied to river course. Figure B.1 shows the synthetic daily variable river discharge that was applied at the inlet of Cosumnes river for 60 days. Figure B.2 –B.4 are the test results.

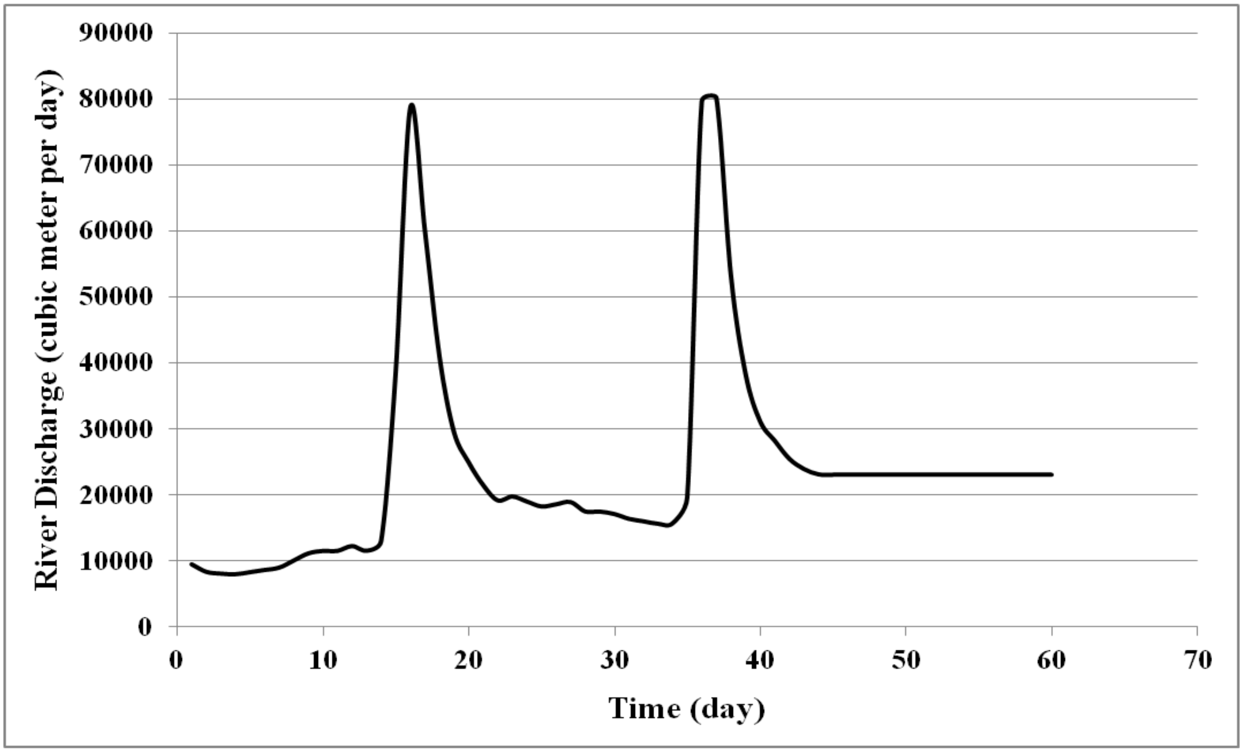


Figure B.1 Synthetic daily variable river discharge.

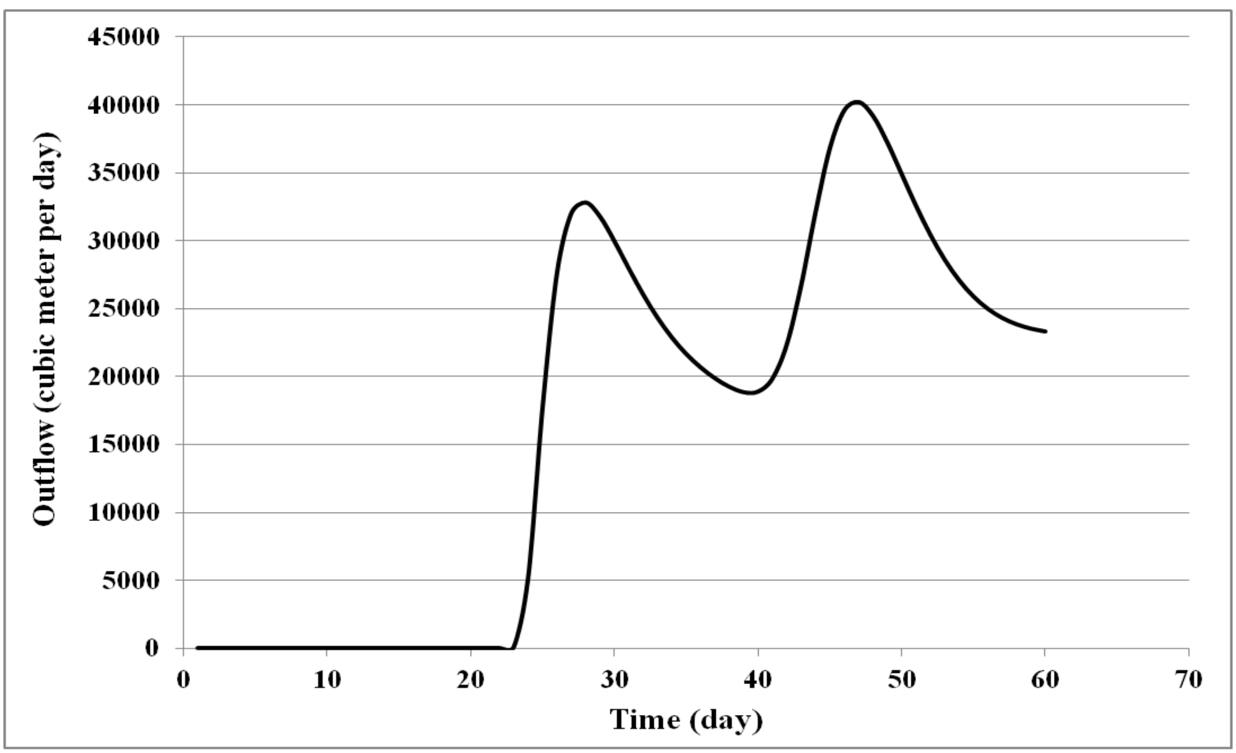


Figure CB.2 Modeled stream flow hydrograph at the outlet of the river.

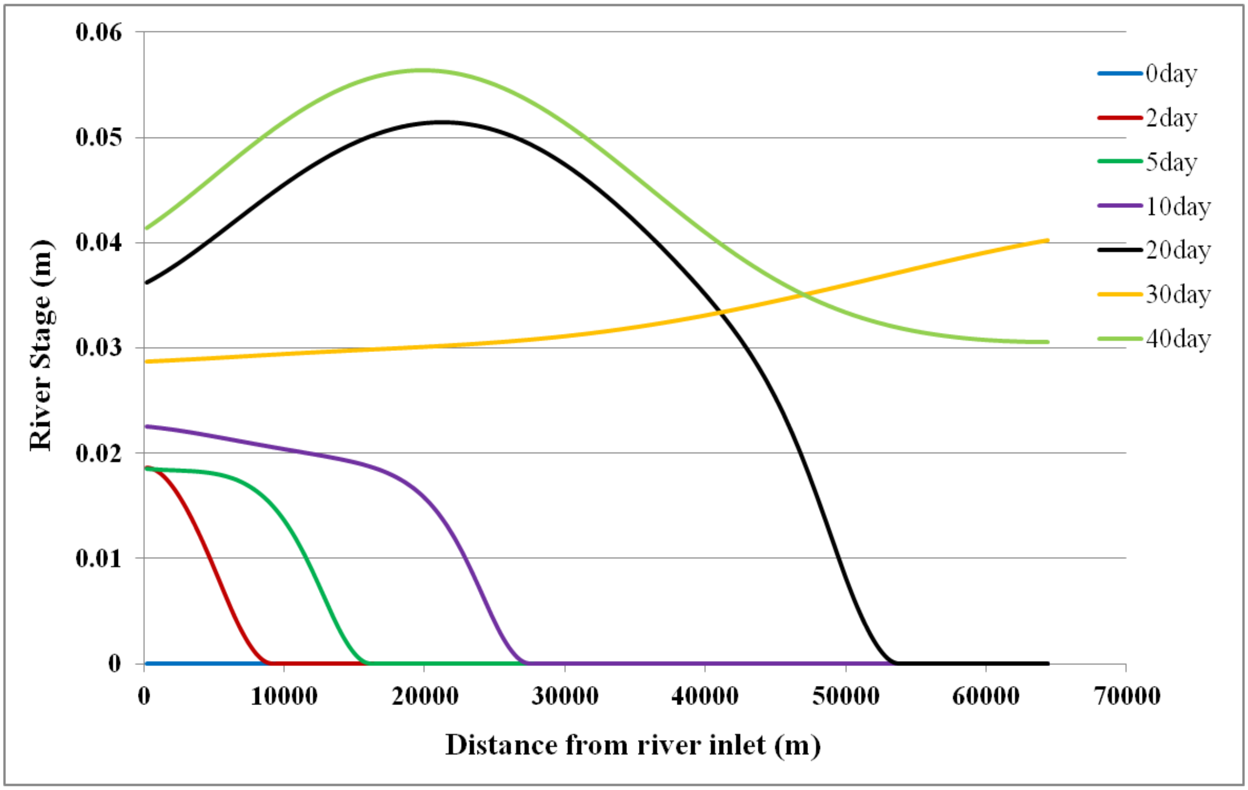
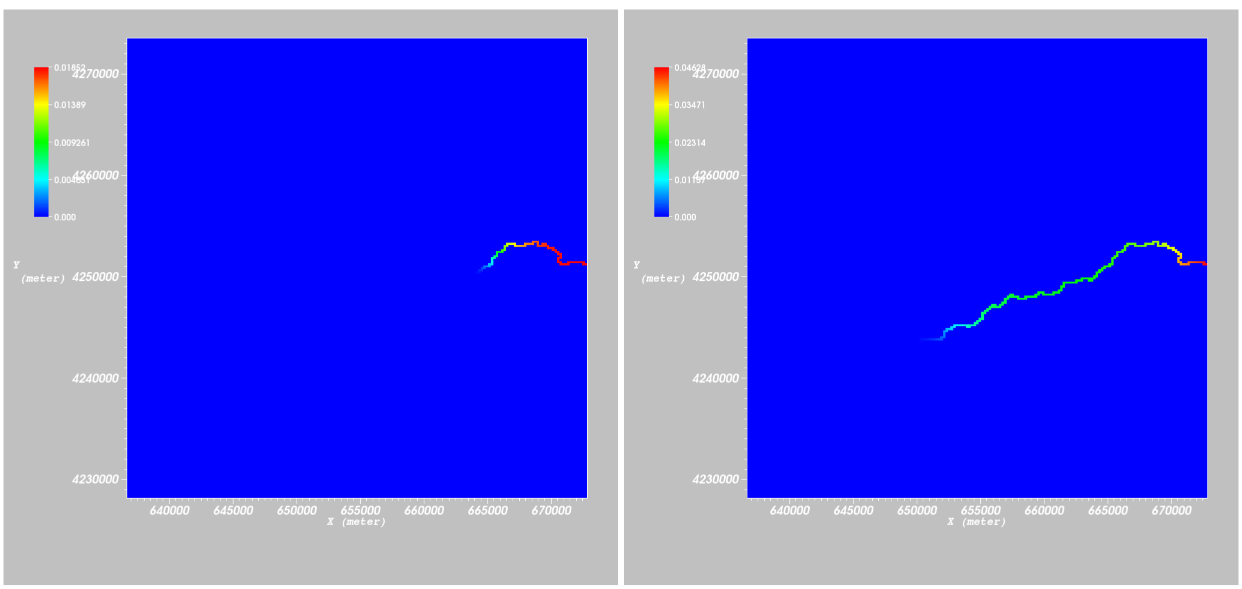


Figure B.3 River stage along river course at various time.



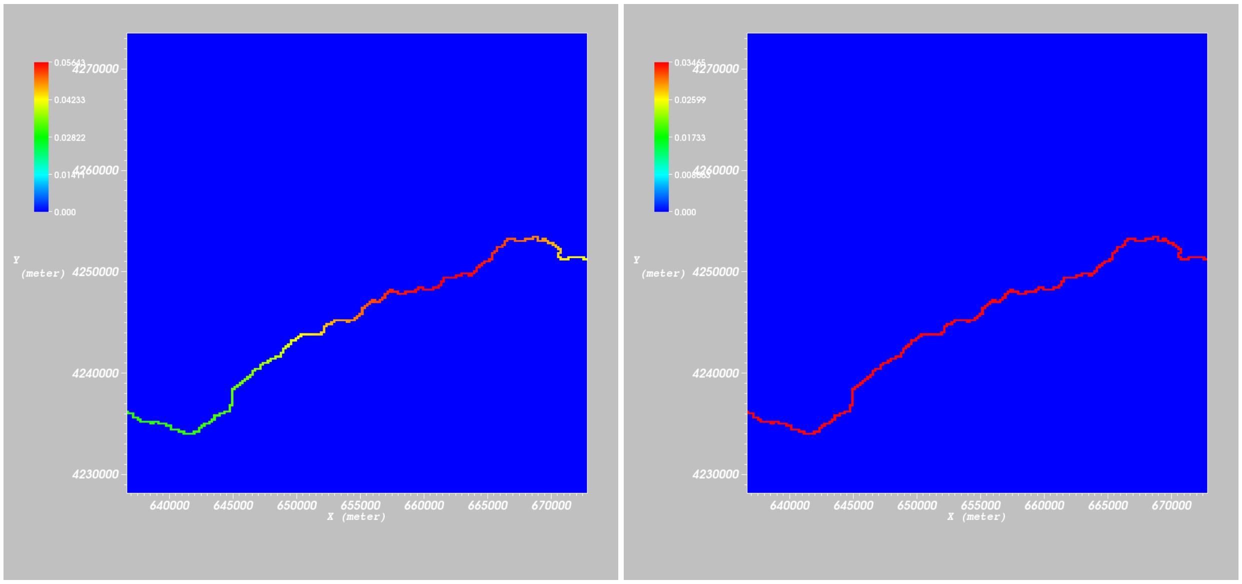


Figure B.4 Pressure distribution along river course at Time 5 day, 15 day, 40 day, 60 day, respectively.

The modeled stream hydrograph in Figure B.3 shows two distinct increments in river stage at time T= 20 days and T=0 days in response to high river discharge event at T=18 day and T=38 day. Runoff at the outlet of the river (Figure B.2) also shows two distinct peaks in response to two high discharge events upstream, but slightly delayed in time. Pressure profile (Figure B.4) evolution along the river course demonstrates that flow routing smoothly from upstream to downstream. Previous river mapping and flow direction enforcement scheme works perfectly in ParFlow, and river response is as expected shown in river hydrograph and runoff.

However, in this modeling effort, overland flow boundary condition (stream routing) is not implemented, due to the very long computing time when including stream routing. Instead, stream stage is applied along river courses as specified head boundary condition for capturing stream recharge.

Appendix C

The American River and Cosumnes River alluvial fan deposits originate from the Sierra Nevada Mountains and extend westward to the Sacramento River and Sacramento-San Joaquin Delta. The depth of the aquifer at the center of the valley is as deep as 2000 feet (600 meters) (RMC 2011). The heterogeneous geological model generated by TPROGS simulation is only 125 meters deep due to limited available geological data. To include deep aquifers systems, 50 homogeneous layers are added to the bottom of the heterogeneous geological model for representing (600-125=475) meters deep aquifers. The upscaling of hydraulic parameters of deep aquifer is as following.

C.1 Equivalent hydraulic conductivity of deep aquifer

With no data available to determine the hydraulic characteristics of the deep aquifer, it is assumed that the hydraulic parameters of the deep aquifer are equivalent to that of the upper heterogeneous aquifer system. The equivalent hydraulic conductivity values of the heterogeneous system were calculated by running so-called "permeameter" simulations in which steady, unidirectional flow was imposed in the x (east-west), y (north-south) and z directions, in three separate simulations, respectively. In these "permeameter" simulations, a unidirectional hydraulic gradient was imposed by specifying heads on two opposite sides of the model, with no flow conditions on all other sides. The model parameters used in "permeameter" simulations are given in Table C.1. The calculated equivalent hydraulic conductivities are listed in Table C.2.

Table C.1 hydraulic parameters of hydro-facies

Hydro-Facies	Hydraulic Conductivity	Specific Storage	Porosity	VG-Alpha	VG-N	VG-Residual Saturation	VG-Saturated Saturation
Gravel	45.26 m/day	2.0e-5	0.35	3.55	3.16	0.1	1.0

Sand	27.12m/day	5.0e-5	0.35	3.55	3.16	0.1	1.0
Muddy Sand	0.1m/day	1.0e-4	0.4	2.69	2.0	0.1	1.0
Mud	0.001m/day	1.0e-3	0.45	1.62	2.0	0.2	1.0

Table C.2 equivalent hydraulic conductivity on horizontal and vertical direction.

Direction	Steady State Flux (cubic meter per day)	Hydraulic Gradient	Cross Section Area (square meters)	Equivalent Hydraulic Conductivity (meter per day)
Horizontal X (East-West)	85464.79	0.001	9761000	8.75
Horizontal Y (North-Sourth)	35146.25	0.001	7783000	4.5
Vertical Z	92432.73	0.01	1643480000	0.0056

Rivers that drain the west slope of the Sierra Nevada Mountains flow roughly east to west, which results in the general east-west orientation of channel deposits. This creates more connected high permeable pathways in the east- west direction as compared to the north-south direction. This explains the higher equivalent hydraulic conductivity obtained on horizontal X direction which is almost two times higher than the equivalent hydraulic conductivity on horizontal Y direction. In the vertical direction, however, the large volumetric fraction of low hydraulic conductivity muddy sand and mud hydro-facies act as natural flow barrier that impedes downward flow, thus the vertical equivalent hydraulic conductivity is determined more by the muddy sands and muds other than other types of sediments. For an extreme case, where the system is homogeneous and contains only mud, the vertical equivalent hydraulic conductivity will be the same as that of mud. Here, in this heterogeneous system, there are vertically connected high permeable pathways, resulting in a higher vertical equivalent hydraulic conductivity than that of mud.

C.2 Transmissivity upscale

Here all calculation is done for the deep homogeneous aquifer only. Anisotropy in hydraulic conductivity is not considered for now. Assuming the horizontal equivalent hydraulic conductivity equals 8.75 meter per day, as calculated above.

Transmissivity is given by (1):

$$T = K_h \cdot D \qquad (1)$$

Where T is transmisivity, K_h is equivalent horizontal hydraulic conductivity, D is the thickness of aquifer.

Here, the transimssivity of upscaled (50 meters thick) and original (475 meters thick) aquifer should equal.

$T_{upscale} = T_{new} \; => 475m * 8.75m/day = 50 * K_{new}$

Then K_{new}=83.17 meter/day

This is the up-scaled hydraulic conductivity for deep aquifer without considering anisotropy.

C.3 Storativity up-scale

Here all the calculation is done for the deep homogeneous aquifer only. Assume, originally, the specific storage of deep aquifer is the same as the specific storage of sand, which is 5.0e-5.

For confined aquifer, the storativity is give as (2)

$$S = S_s \cdot D \qquad (2)$$

Where S is aquifer storativity, S_s is specific storage, D is the thickness of aquifer.

Here, the storativity of upscaled (50 meters thick) and original (475 meters thick) aquifer should equal.

$S_{old} = S_{new} => \ 5.0e - 5 * 475 = 50 * S_{snew}$

Then, S_{snew}=4.8e-4.

This is the up-scaled specific storage for deep homogeneous aquifer.

Appendix D

D.1 Calibration-system performance

In this study, the geological model used is one realization of the stochastic TProGS model, which represents only one possible geological structure among many other possibilities. Thus there is uncertainty associated with the geological model and, in turn, the flow model. Therefore, it is highly unlikely that the simulated groundwater head will match with the measured groundwater head at any specified location (such as those where the DWR monitor well locates). Instead, the focus of calibration is the overall performance of the model. The first calibration goal is to see whether the model vertical connectivity or vertical hydraulic conductivity is consistent with field conditions. The secondary calibration goal is to examine whether the simulated groundwater levels in space and time are broadly consistent with field observations. The primary goal stems from the model objectives and the origins of the pumpage and recharge data used to drive the current model. Recall that the SACIWRM model (RMC 2011) is based on the conventional practice of using relatively homogenous aquifer properties, with anisotropic hydraulic conductivity values to represent the restricted connectivity or confinement caused by the aquitard layers that are explicitly included in the heterogeneous model used in this study. And the pumping and recharge in SACIWRM model was estimated by conventional crop consumptive use method. Because the goal of the model is to investigate floodplain recharge processes rather than to make specific predictions, this calibration is first to answer the question "Is the heterogeneous model consistent with the field observations, and if not, can modest adjustments in hydro-facies hydraulic conductivity values and/or in the pumpage numbers make the model consistent?"

To evaluate the first calibration goal, 210 hypothesized monitoring wells were randomly sampled, which spread evenly onto the surface of the model. Both shallow semi-confined and deep confined groundwater head are sampled. The vertical distance between "shallow" and "deep" aquifers is at least 30 meters to insure they do not sample the same aquifer. The deep aquifer response in model is compared to DWR monitoring well data, because the DWR monitoring wells are sampling mostly deep aquifers. The shallow aquifer response in model is compared with Cosumnes River Group (CRG)) monitoring wells a at the Cosumnes River study site where the monitoring wells are mostly shallow (Personal Communication, 2014, Nick Newcomb) (Figure D.3). The schemes of comparing are summarized as following: for a selected monitoring well, search the randomly sampled well nearby. If similar groundwater head pattern and seasonal variation is observed in both field observations and in model simulation, then it is convincing that the model is consistent with field conditions. Remember the geological model is one realization of TPROGS simulation, it is unlikely a good match in both general head pattern and seasonal variation will be found at all locations. Comparisons of deep groundwater head response are listed in Figure D.1A-X. Comparisons of shallow groundwater head response are listed in Figure D.2A-I.

Comparison of the shallow semi-confined seasonal variation and the deep confined seasonal variation, it is seen that the shallower response is more gentle than the deeper groundwater response, thus the shallow to deep groundwater head pattern is preserved. Therefore the overall model performance is consistent with the field observations.

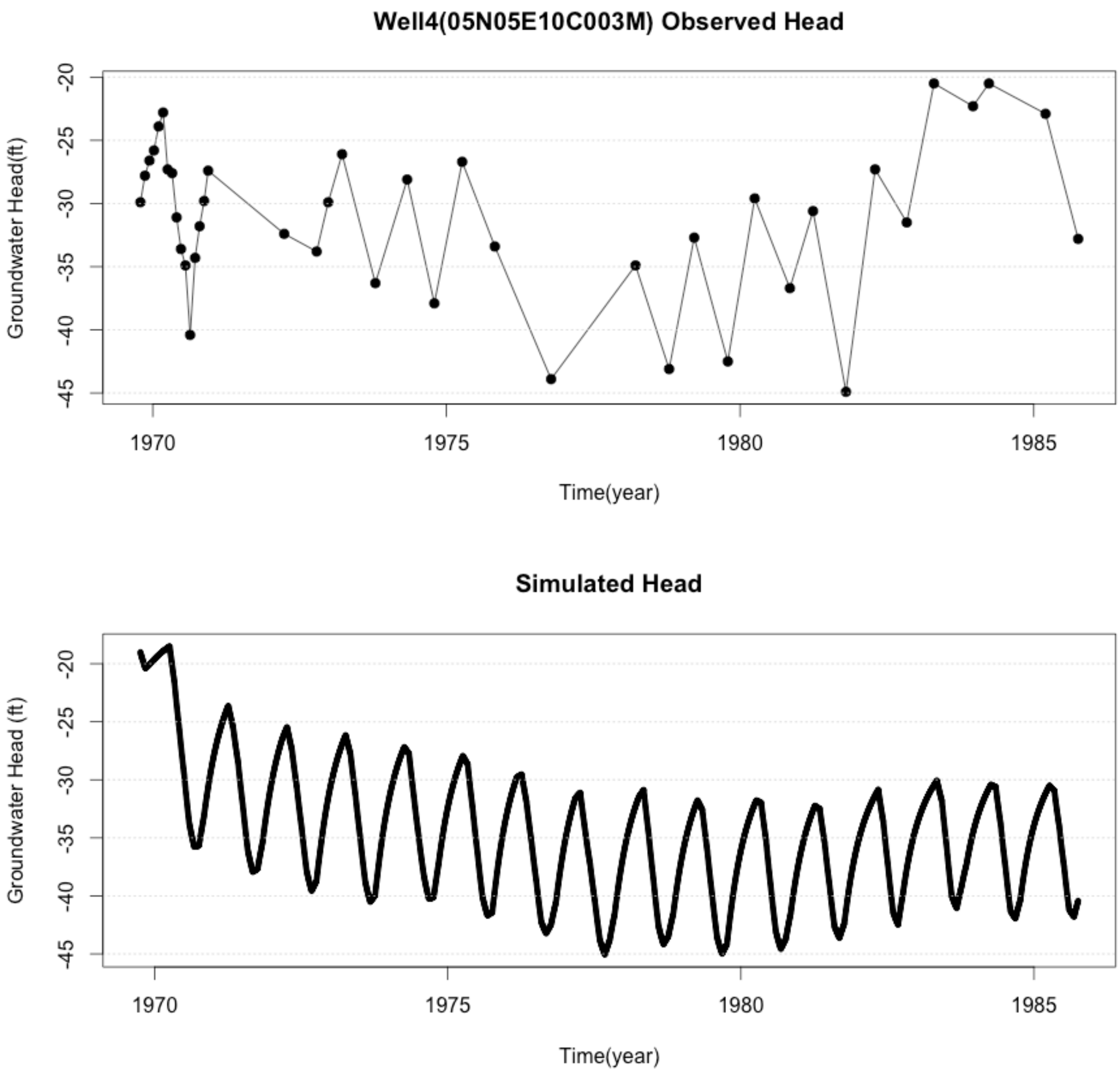


Figure D.1A comparison of simulated and observed deep confined groundwater head at well4(05N05E10C003M)

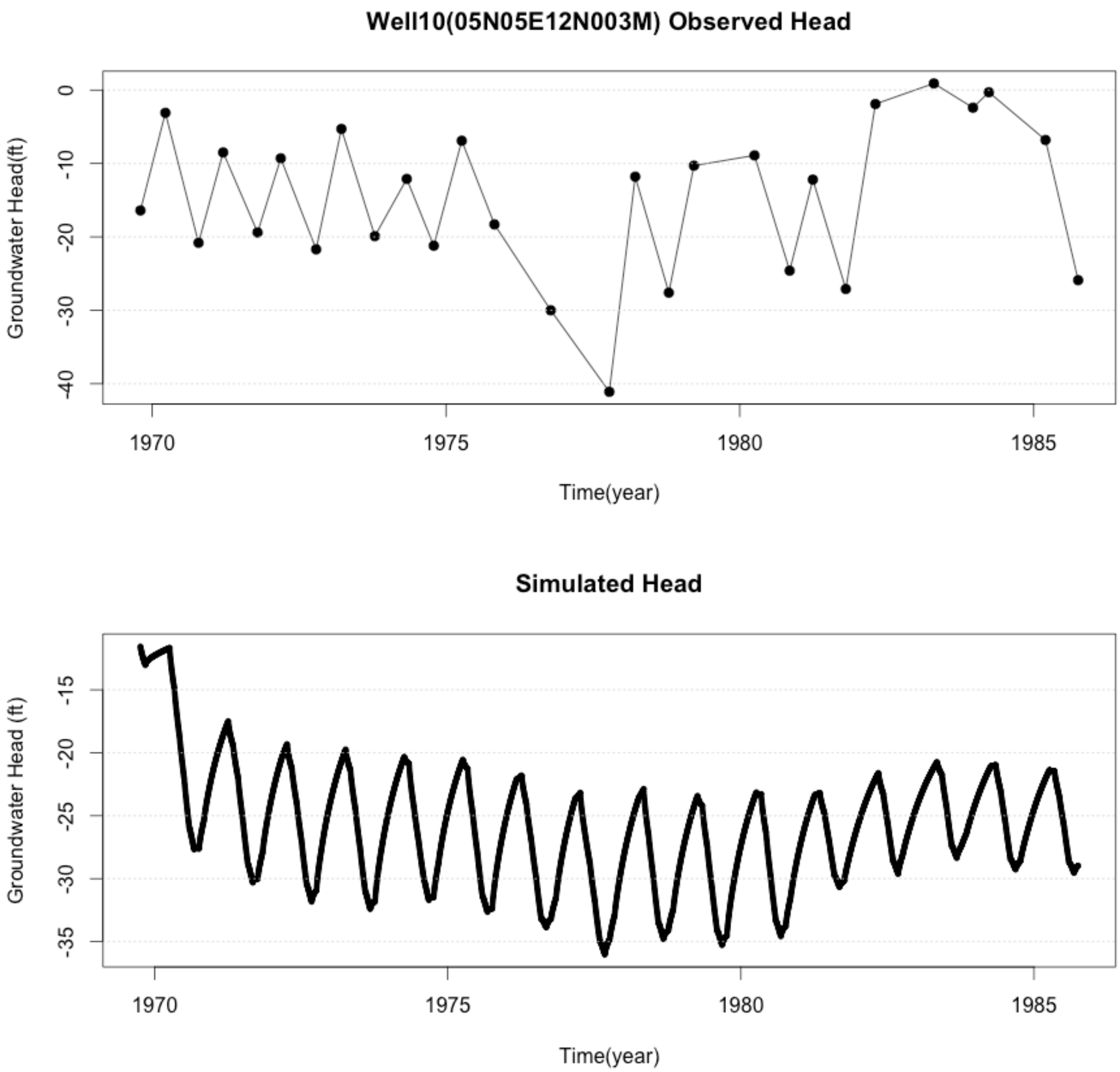


Figure D.1B comparison of simulated and observed deep confined groundwater head at well10(05N05E12N003M)

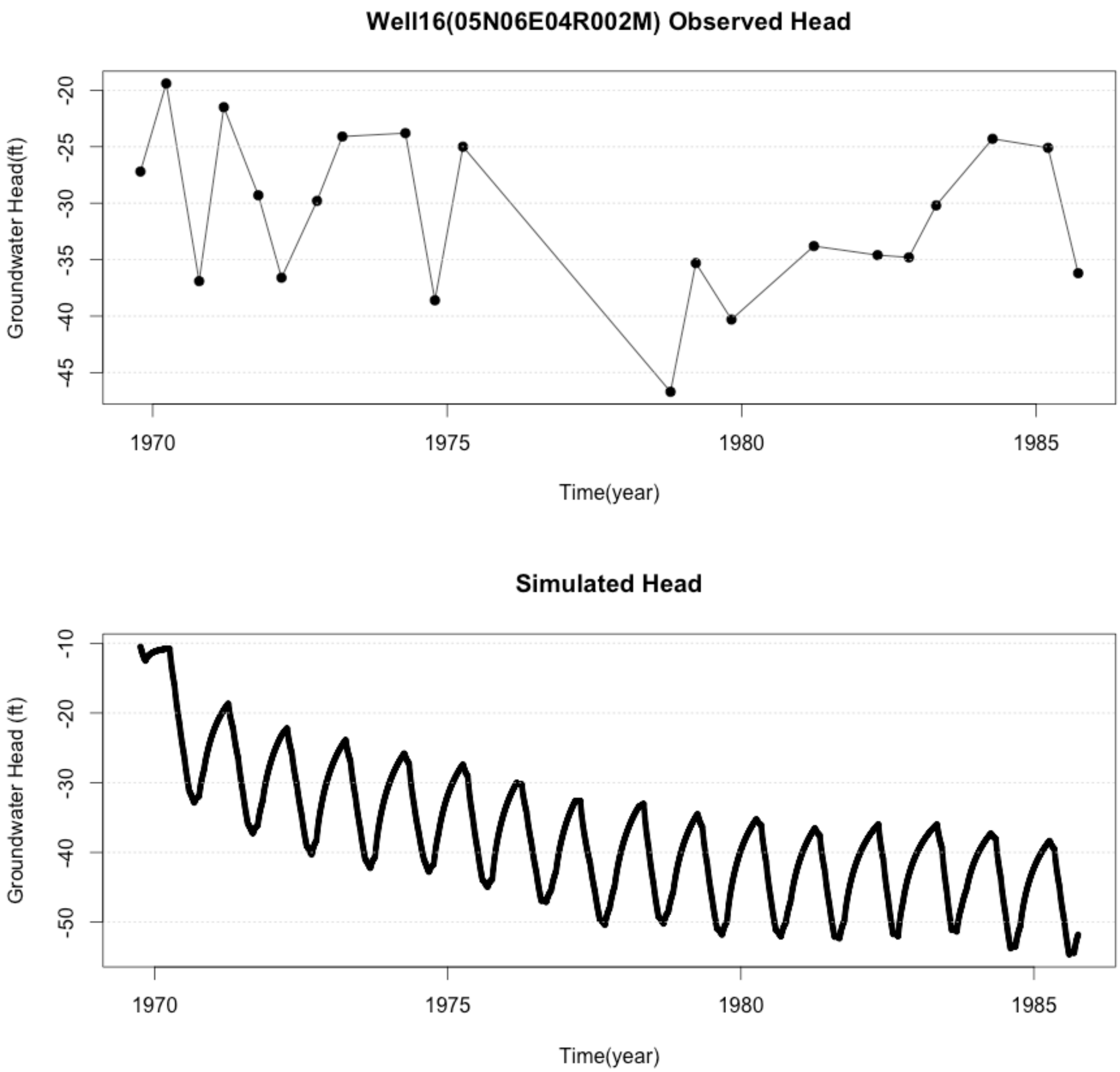


Figure D.1C comparison of simulated and observed deep confined groundwater head at well16(05N06E04R002M)

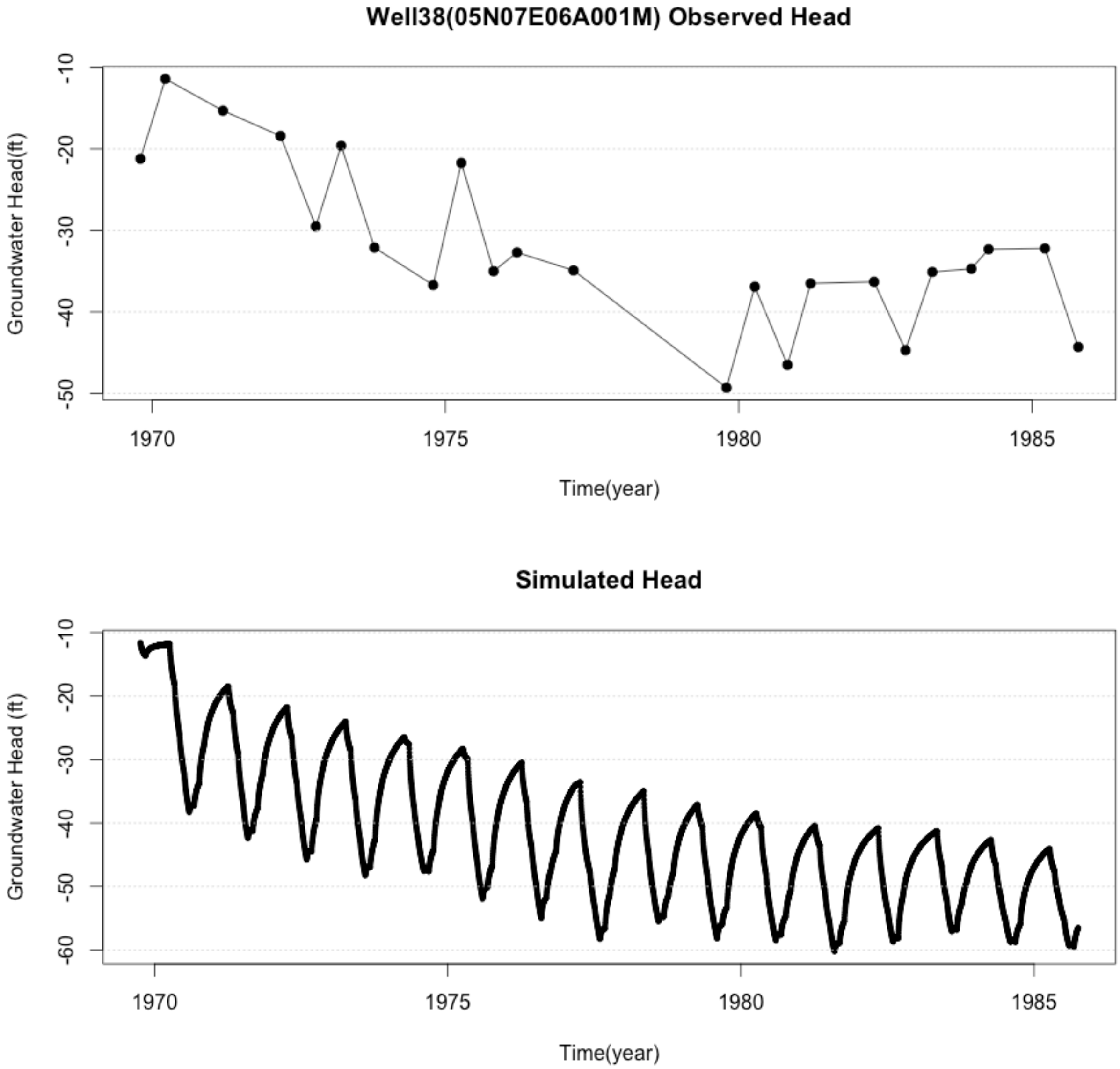


Figure D.1D comparison of simulated and observed deep confined groundwater head at well4(05N07E06A001M)

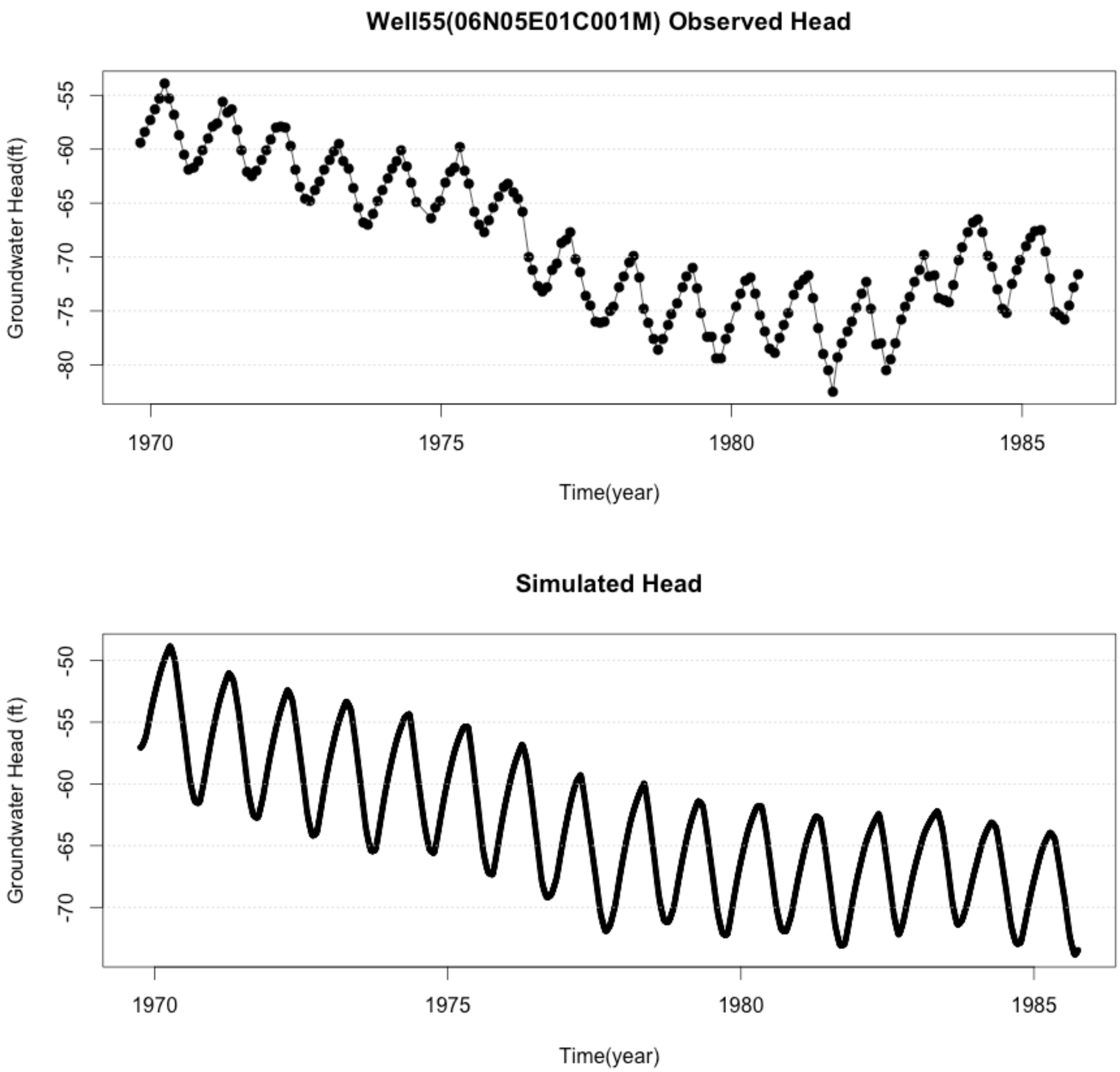


Figure D.1E comparison of simulated and observed deep confined groundwater head at well55(06N05E01C001M)

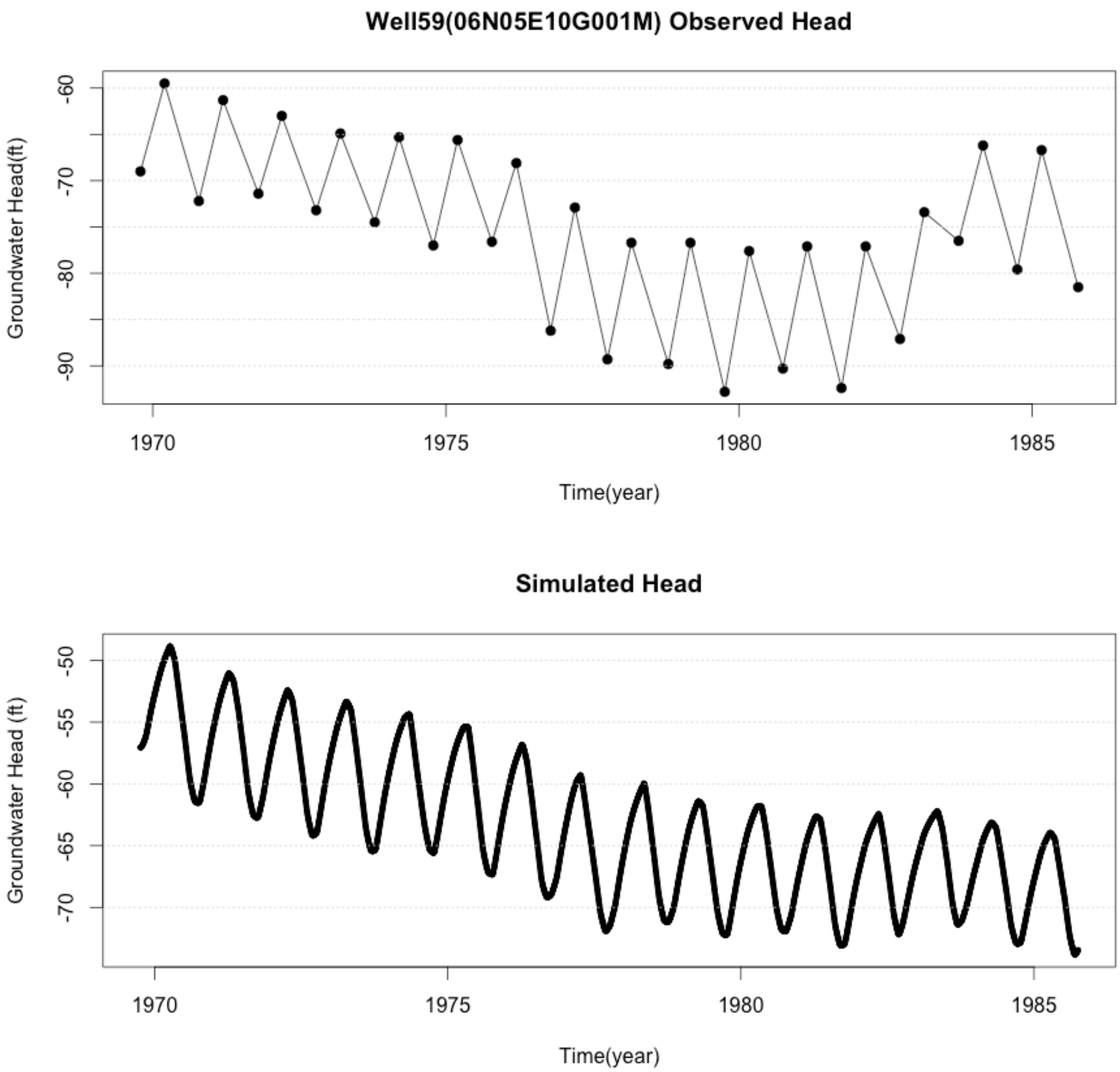


Figure D.1F comparison of simulated and observed deep confined groundwater head at well59(06N05E10G001M)

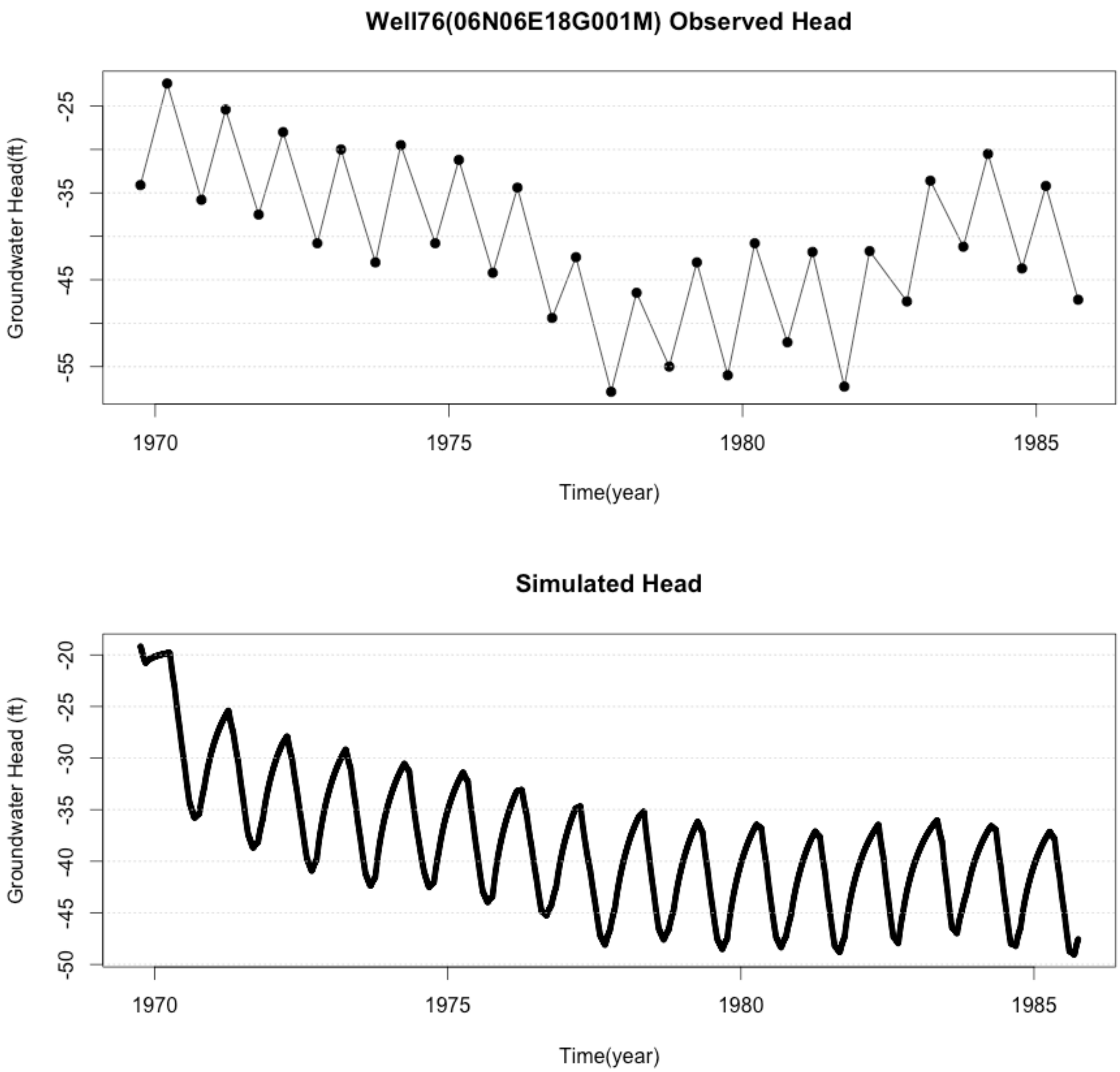


Figure D.1G comparison of simulated and observed deep confined groundwater head at well76(06N06E18G001M)

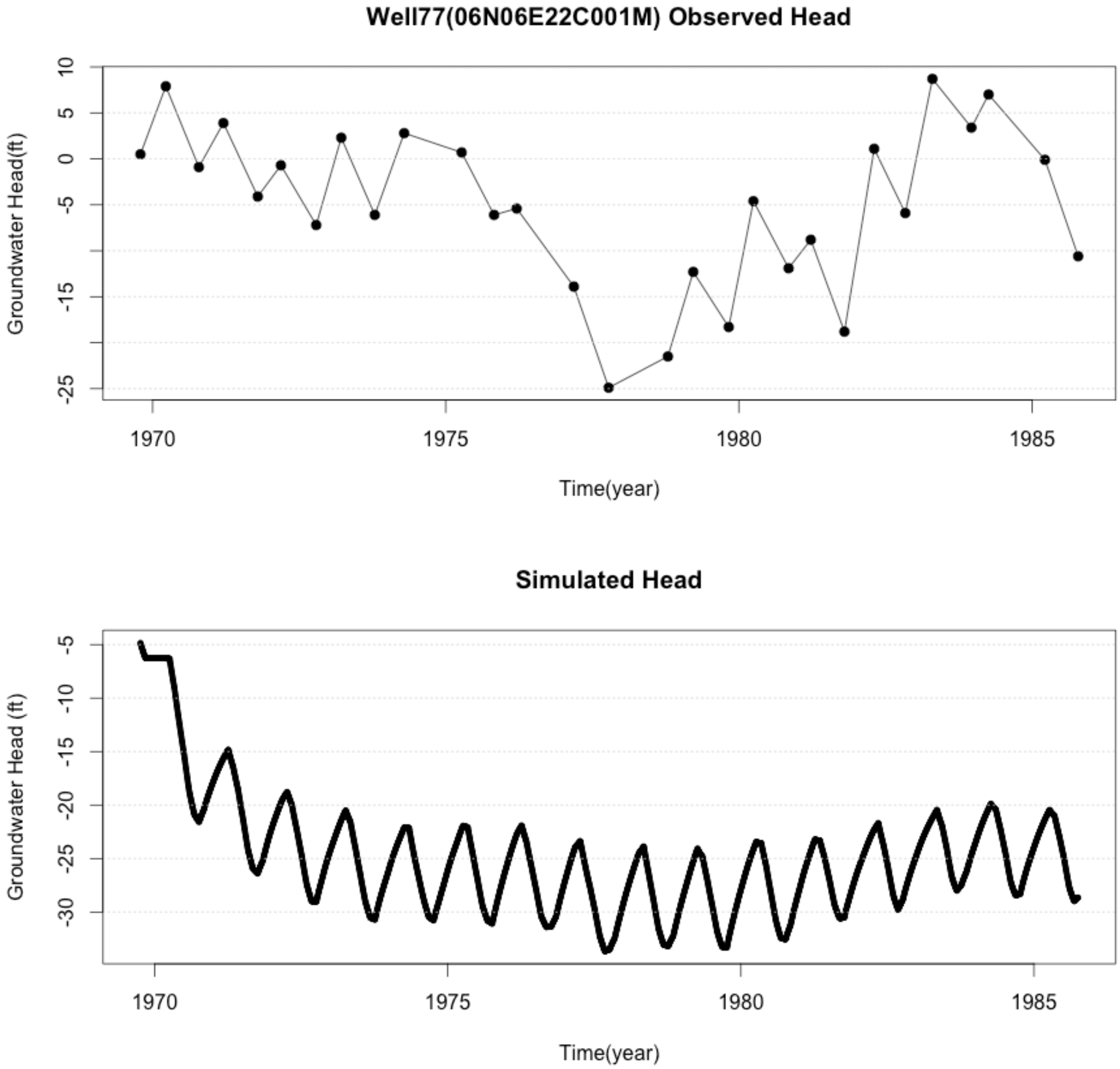


Figure D.1H comparison of simulated and observed deep confined groundwater head at well77(06N06E22C001M)

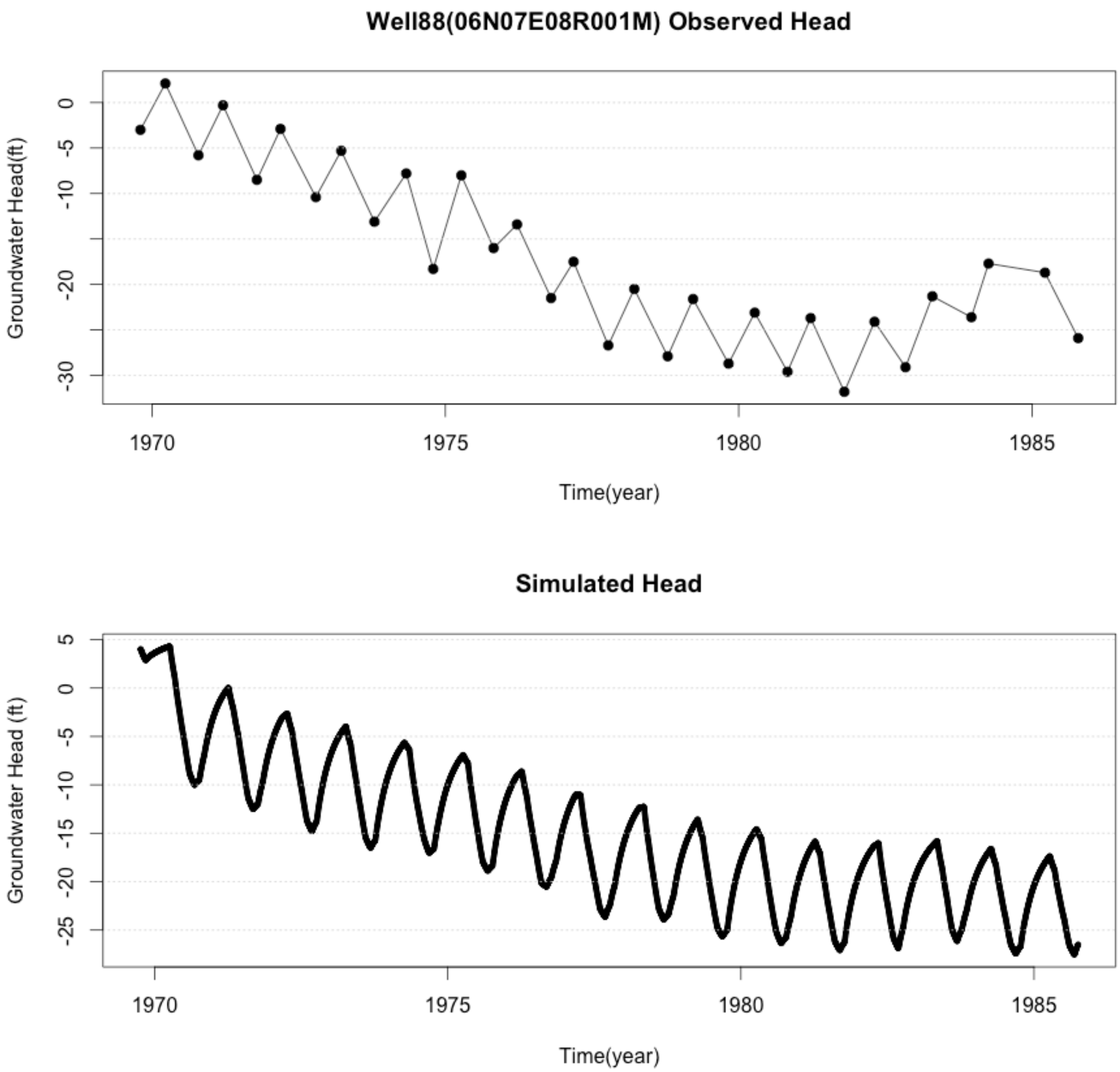


Figure D.1I comparison of simulated and observed deep confined groundwater head at well88(06N07E08R001M)

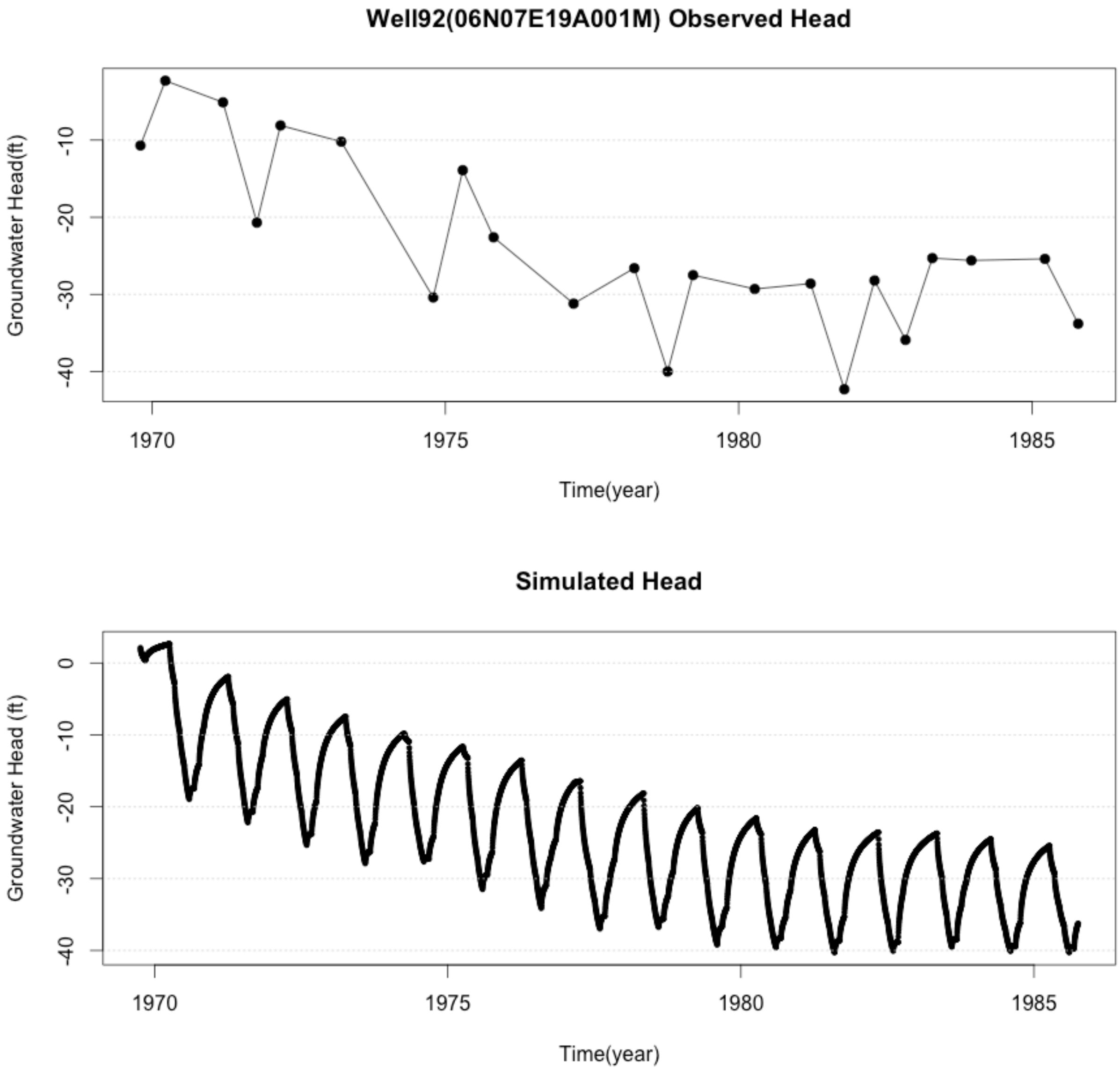
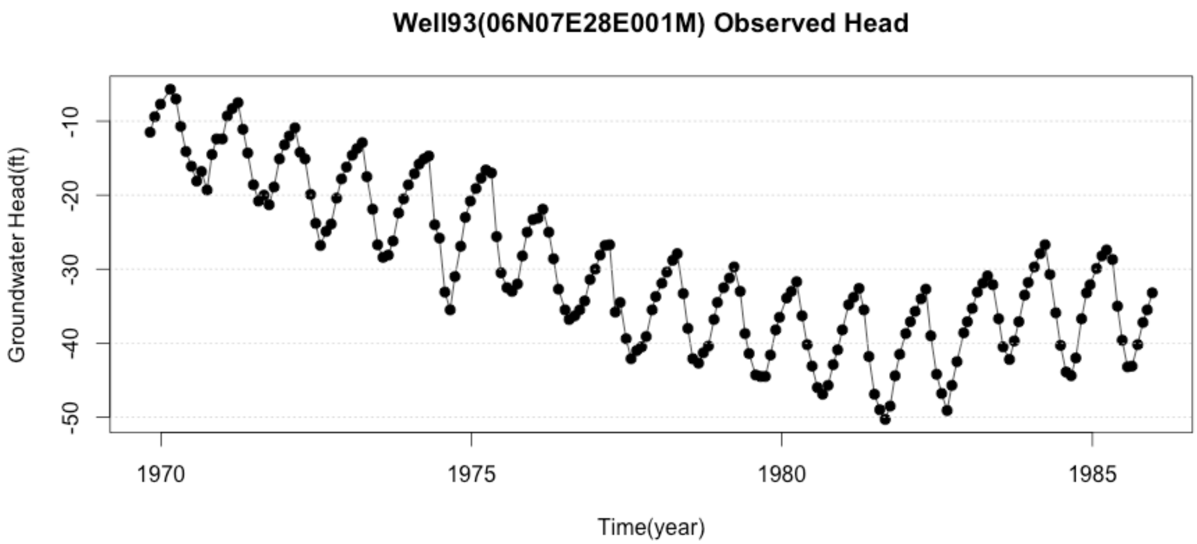


Figure D.1J comparison of simulated and observed deep confined groundwater head at well92(06N07E19A001M)



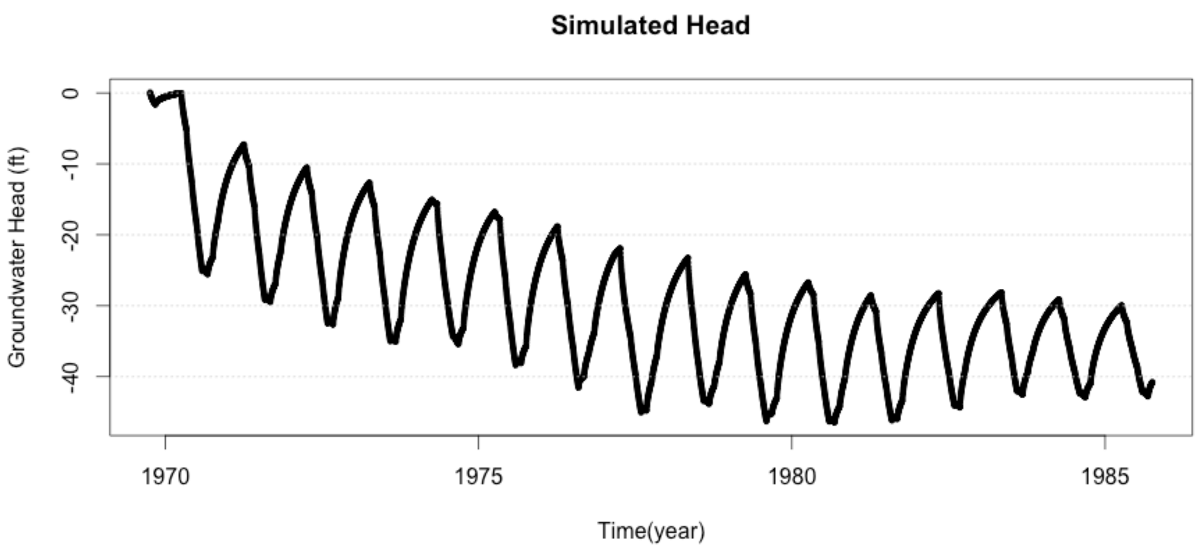


Figure D.1K comparison of simulated and observed deep confined groundwater head at well93(06N07E28E001M)

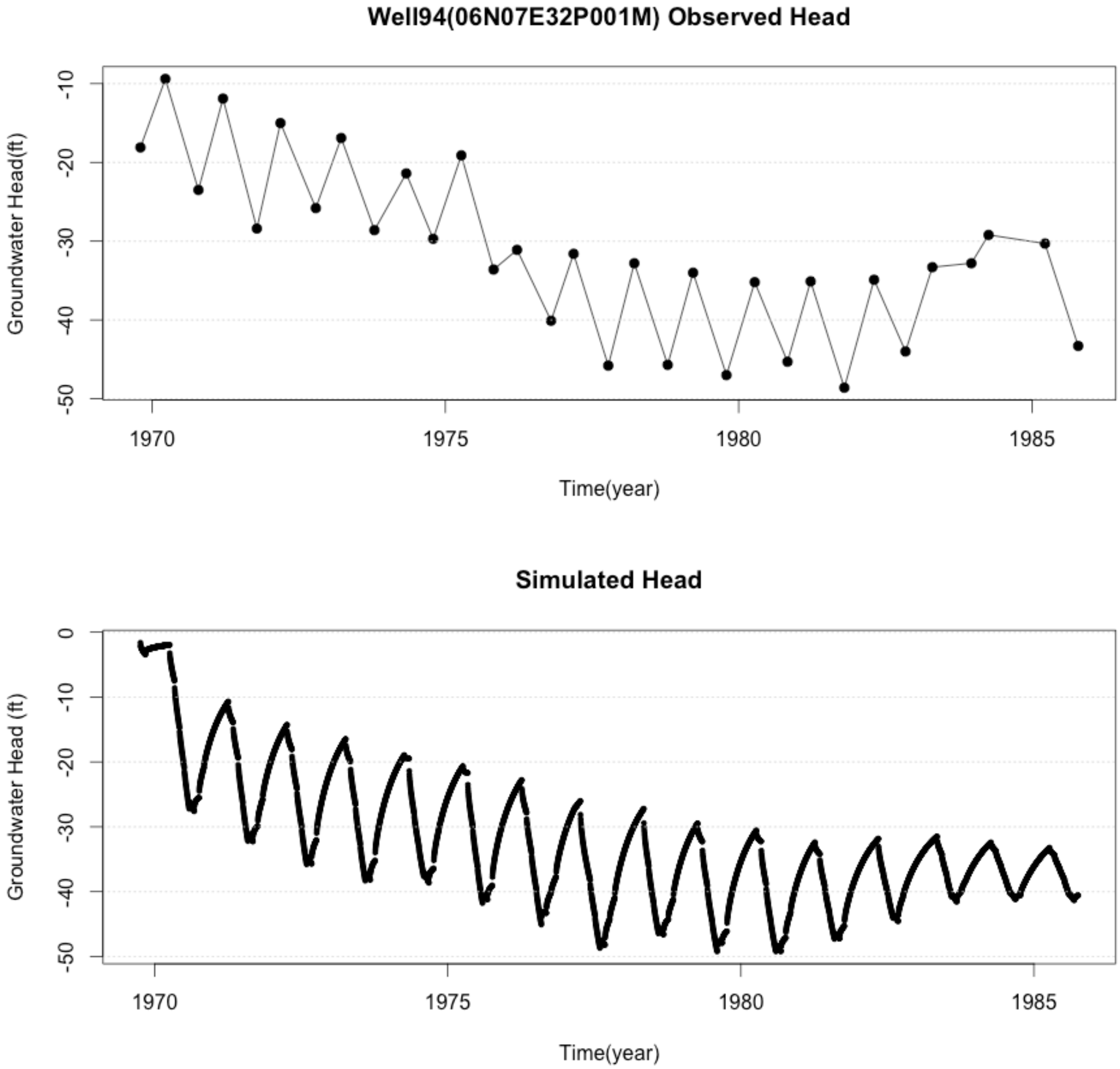


Figure D.1L comparison of simulated and observed deep confined groundwater head at well94(06N07E32P001M)

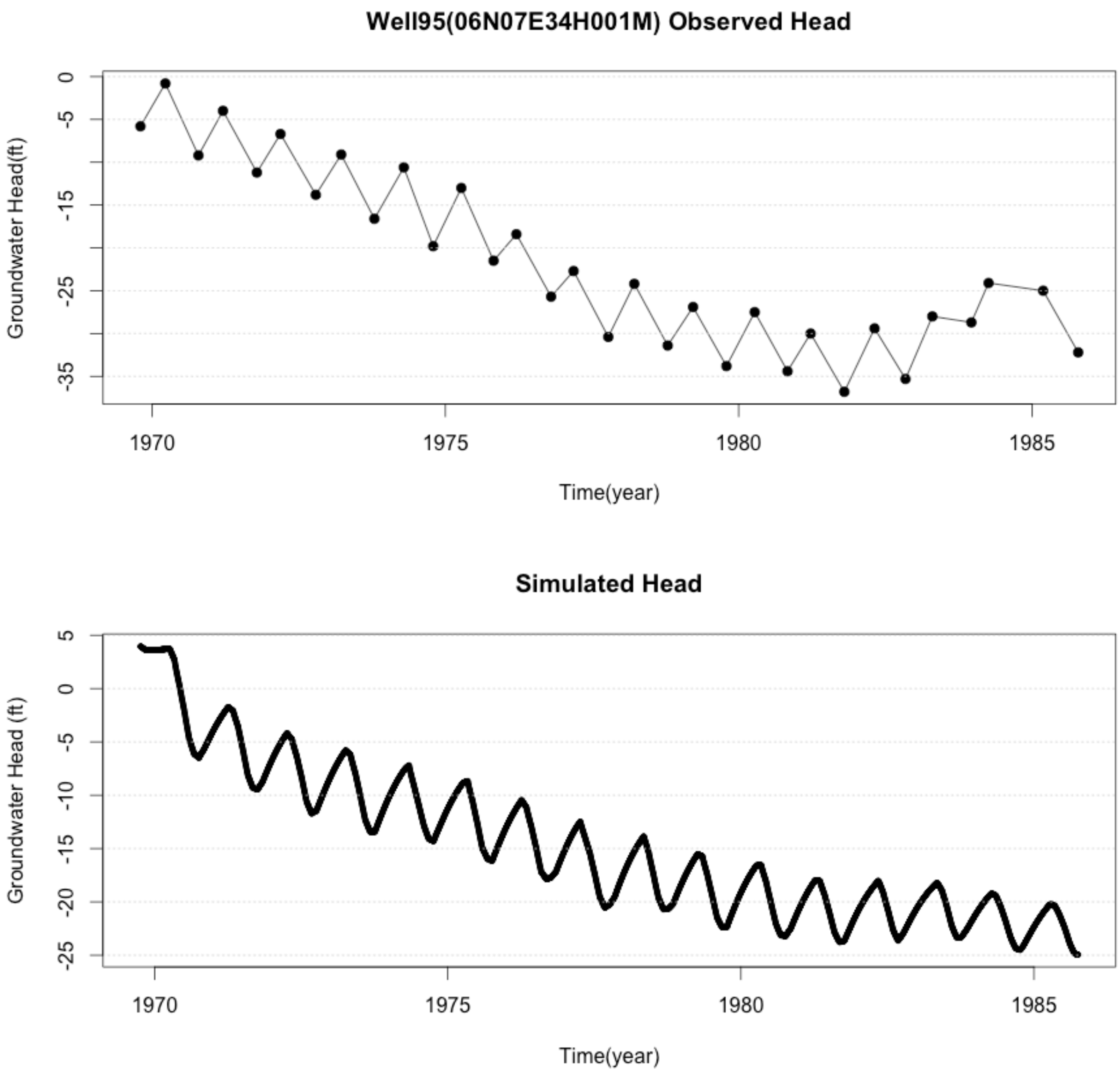


Figure D.1M comparison of simulated and observed deep confined groundwater head at well95(06N07E34H001M)

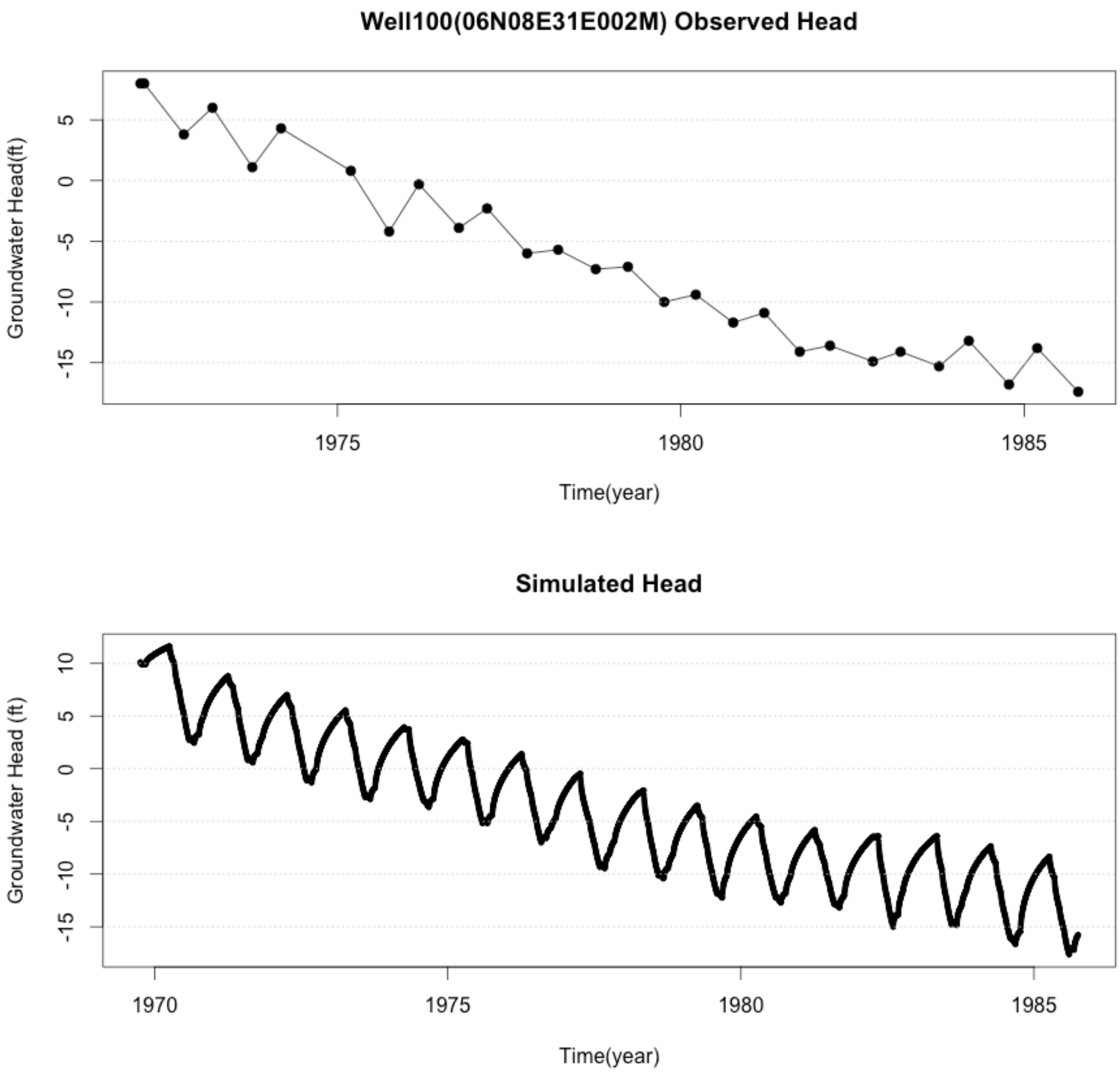


Figure D.1N comparison of simulated and observed deep confined groundwater head at well100(06N08E31E002M)

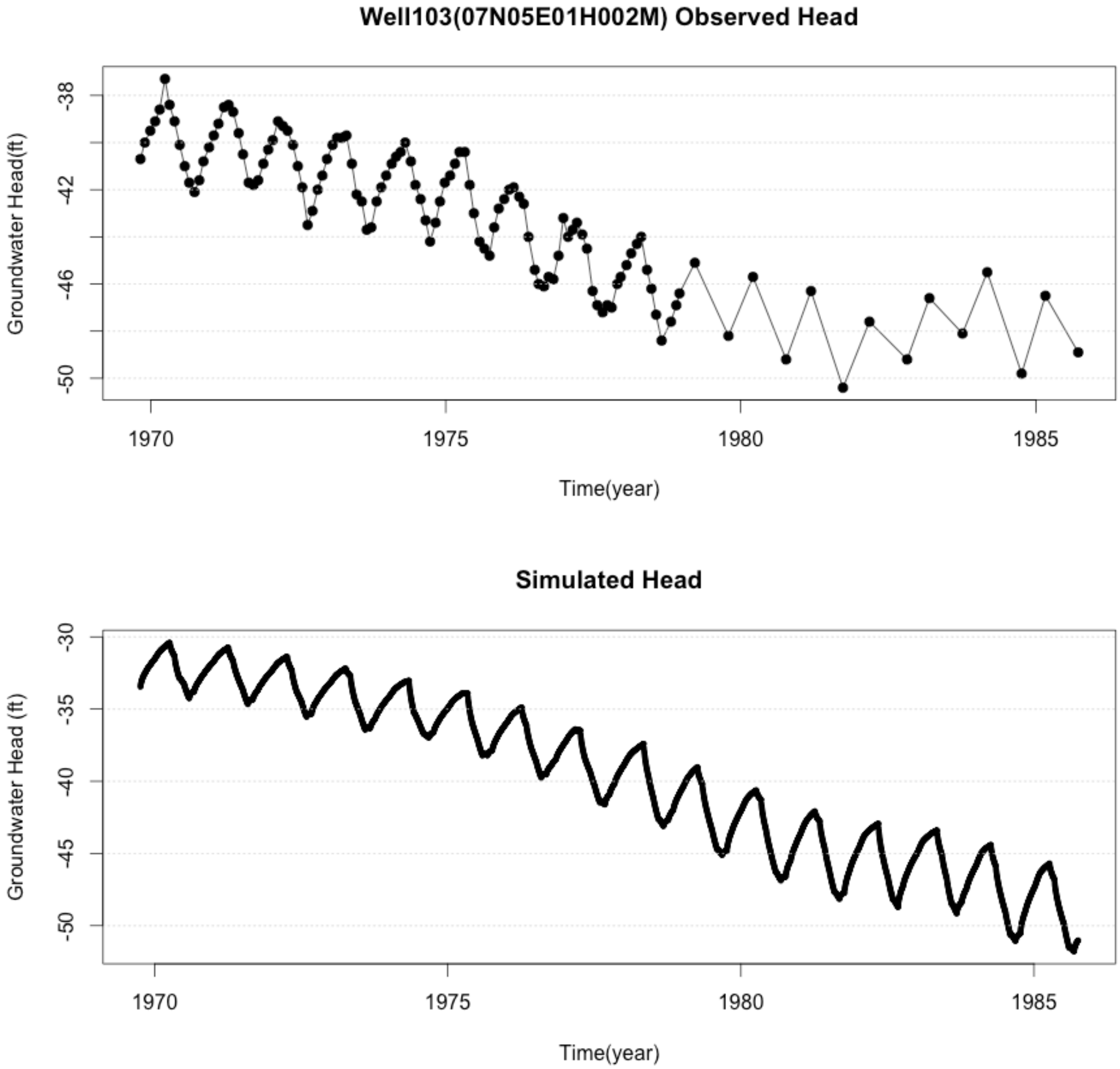


Figure D.1O comparison of simulated and observed deep confined groundwater head at well103(07N05E01H002M)

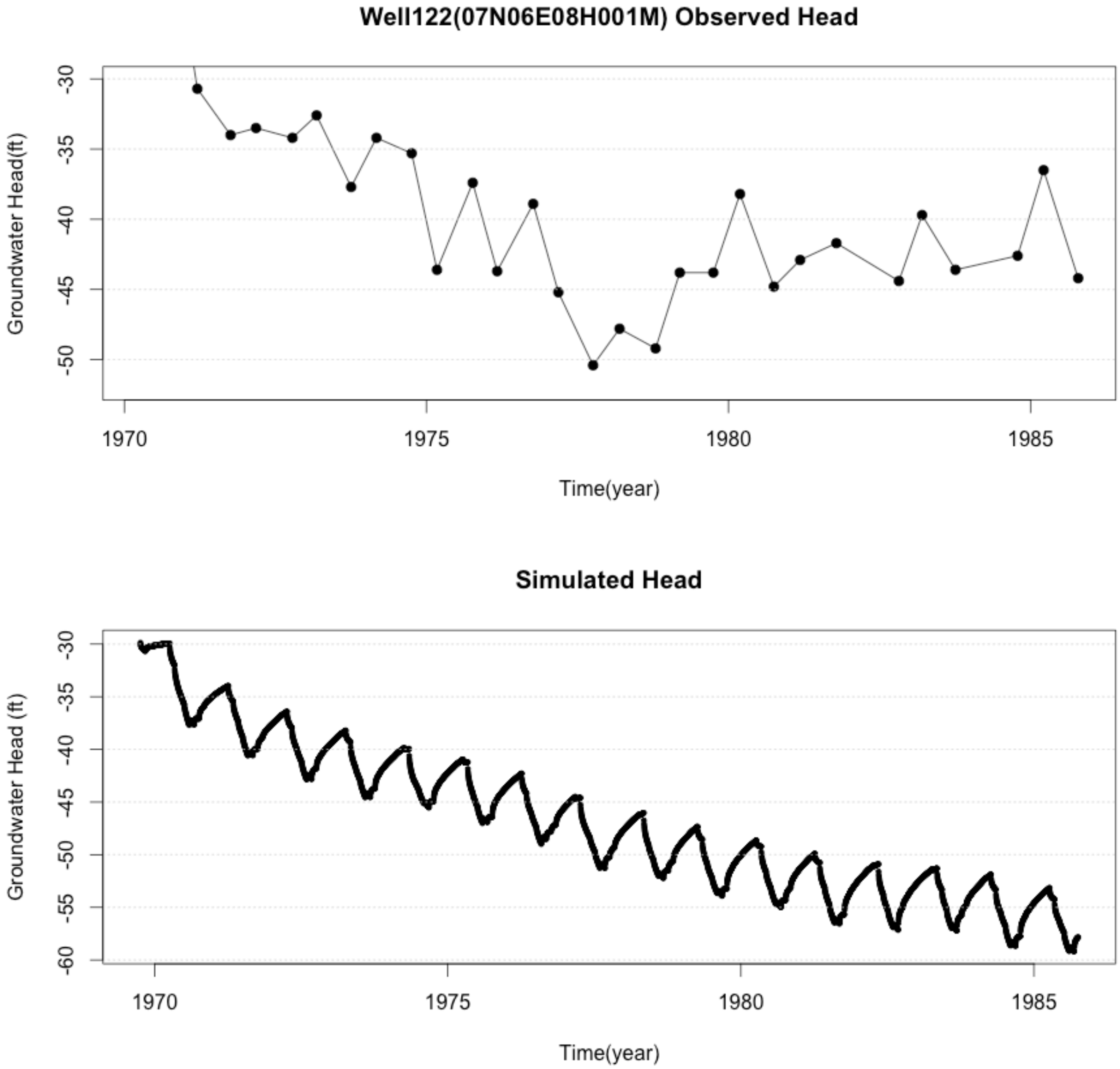
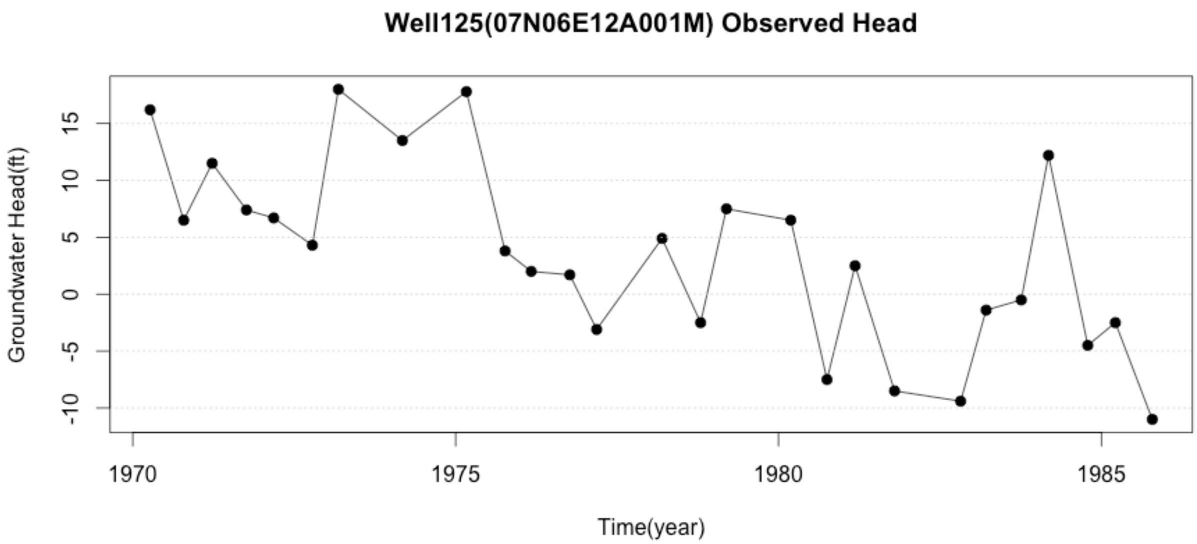


Figure D.1P comparison of simulated and observed deep confined groundwater head at well122(07N06E08H001M)



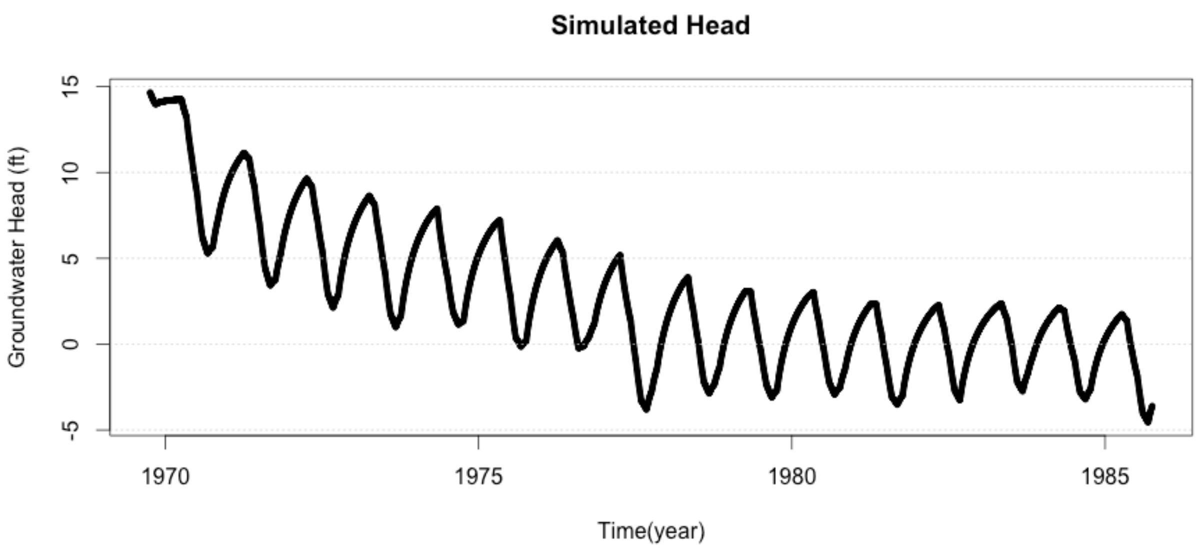
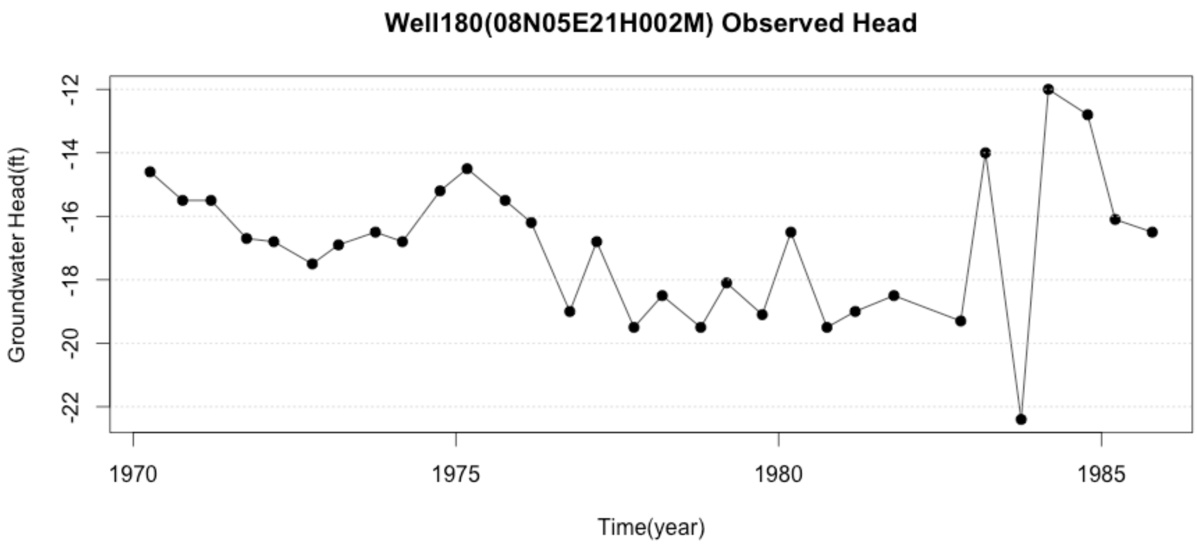


Figure D.1Q comparison of simulated and observed deep confined groundwater head at well125(07N06E12A001M)



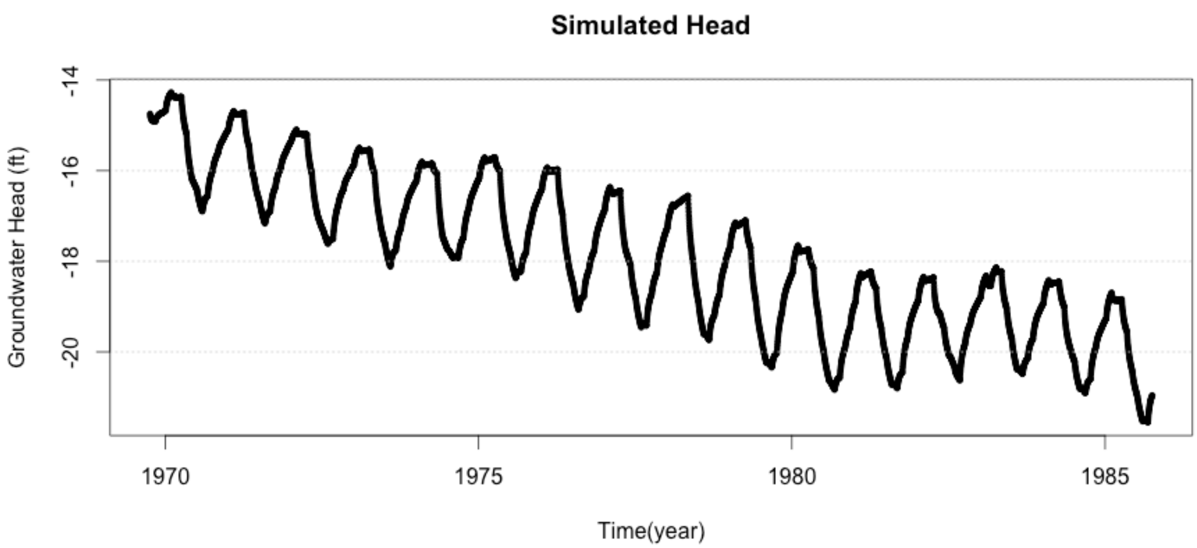


Figure D.1R comparison of simulated and observed deep confined groundwater head at well180(08N05E21H002M)

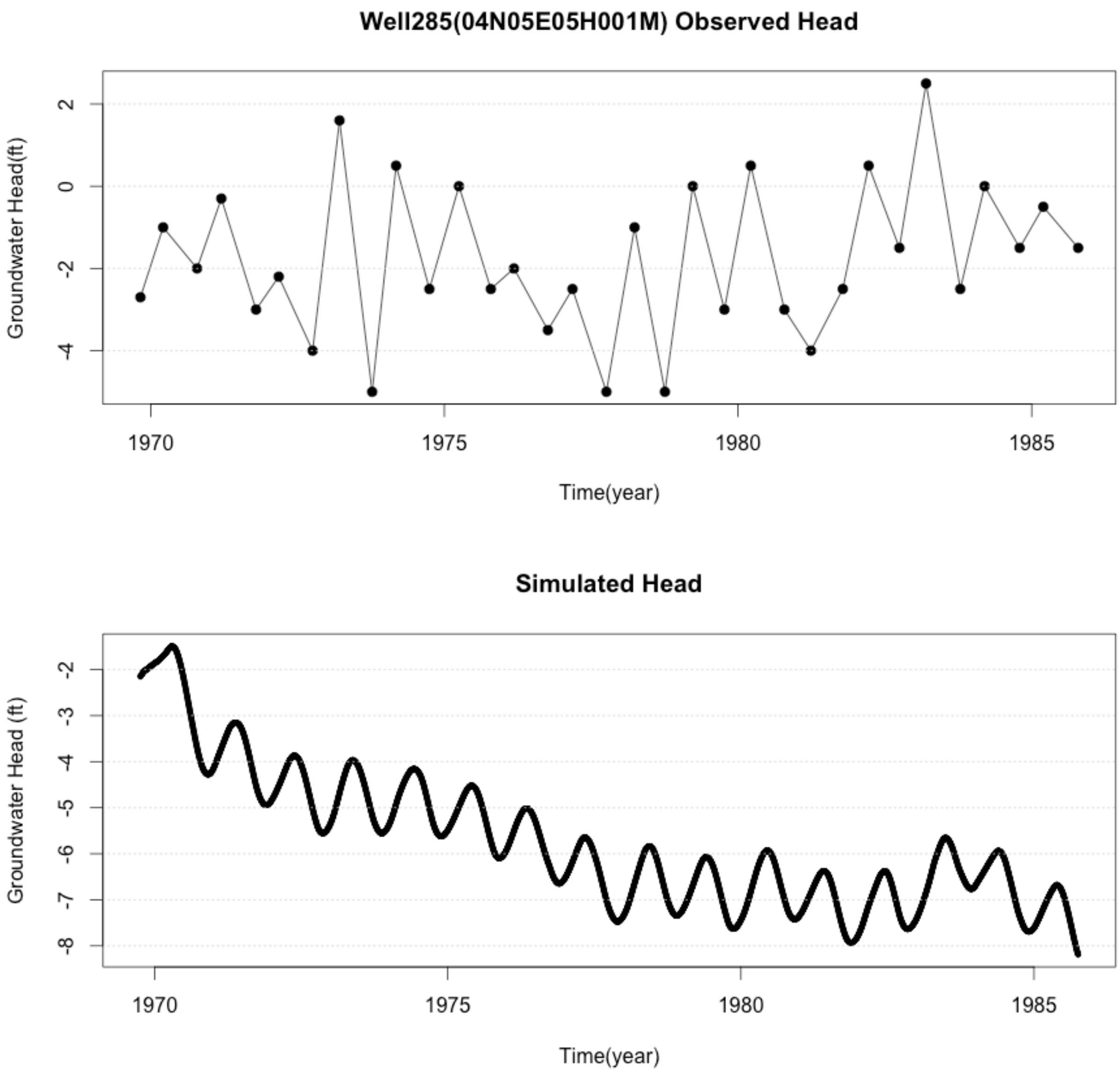


Figure D.1S comparison of simulated and observed deep confined groundwater head at well285(04N05E05H001M)

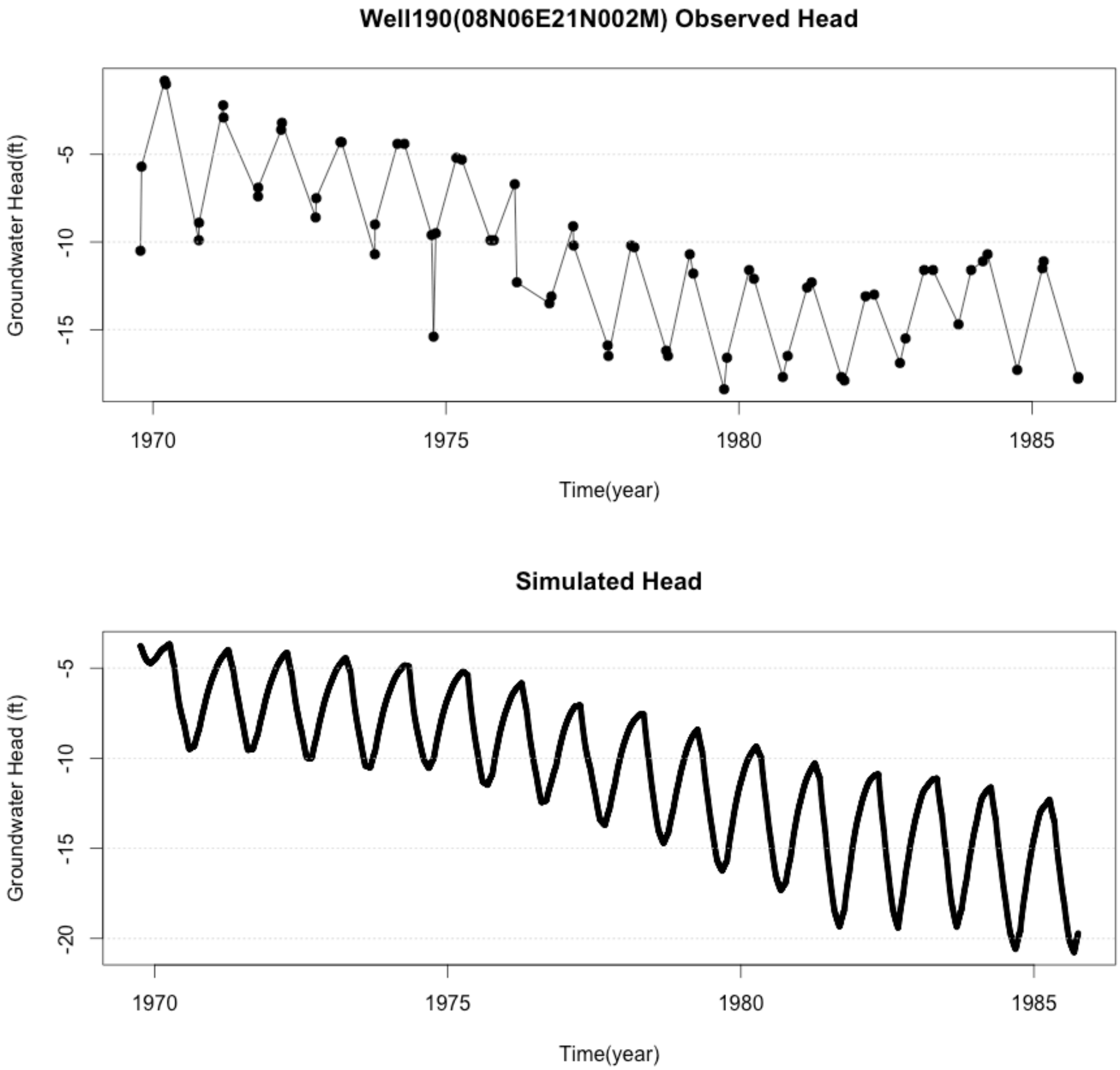


Figure D.1T comparison of simulated and observed deep confined groundwater head at well190(08N06E21N002M)

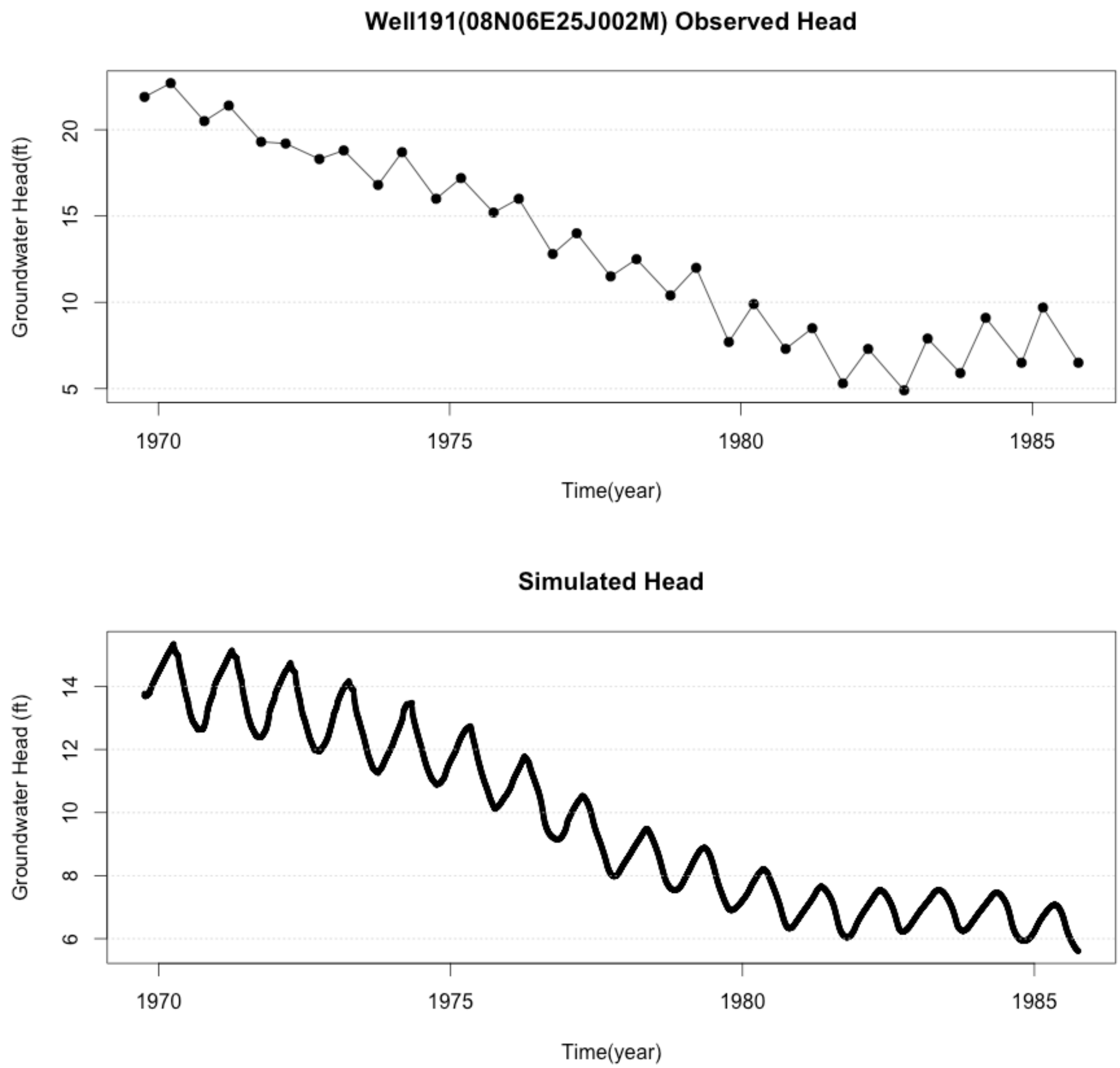


Figure D.1U comparison of simulated and observed deep confined groundwater head at well4(08N06E25J002M)

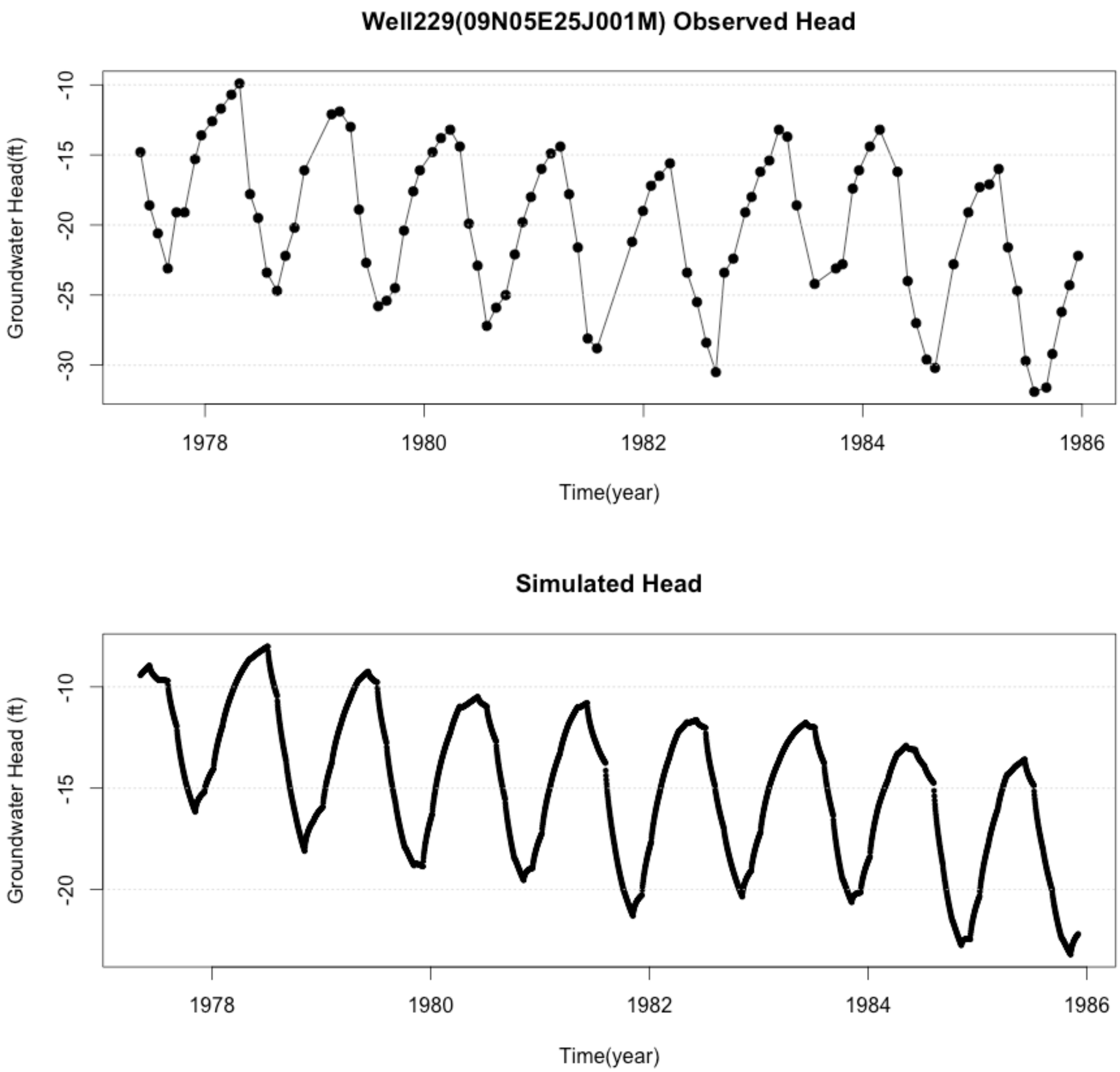
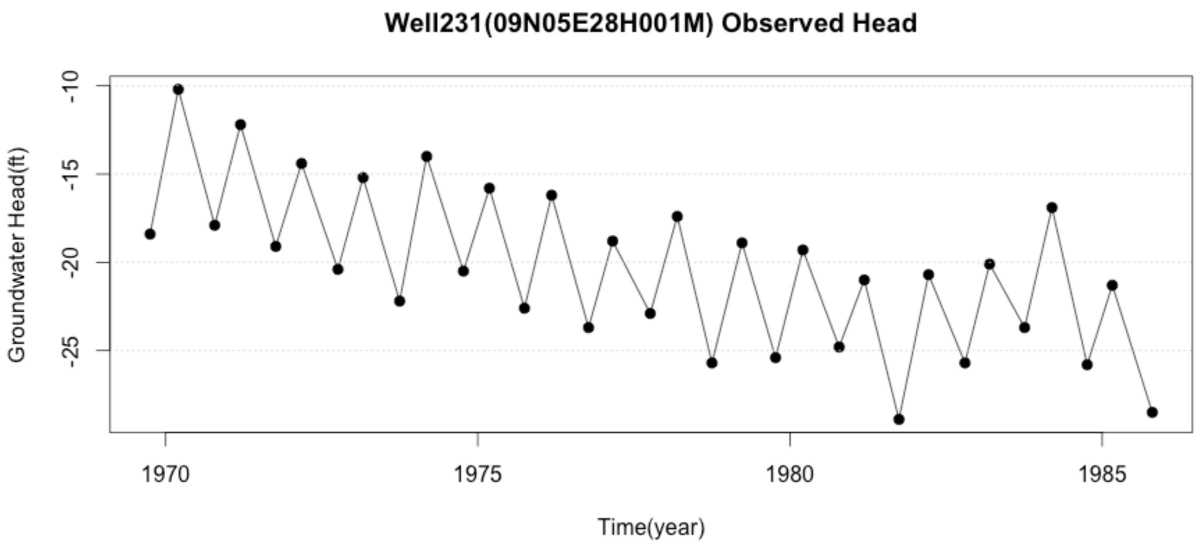


Figure D.1V comparison of simulated and observed deep confined groundwater head at well229(09N05E25J001M)



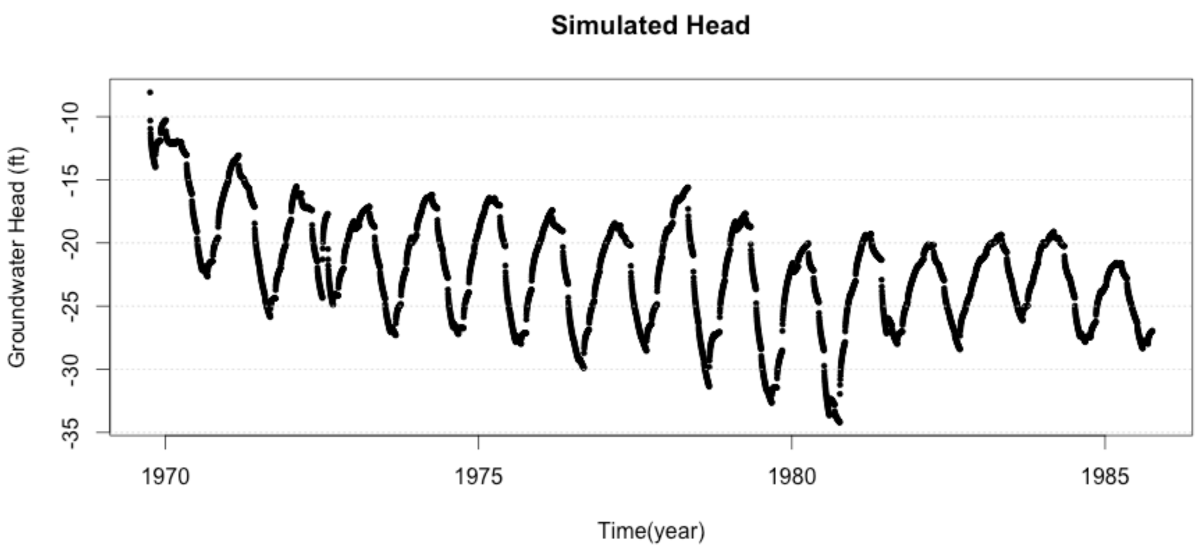


Figure D.1V comparison of simulated and observed deep confined groundwater head at well231(09N05E28H001M)

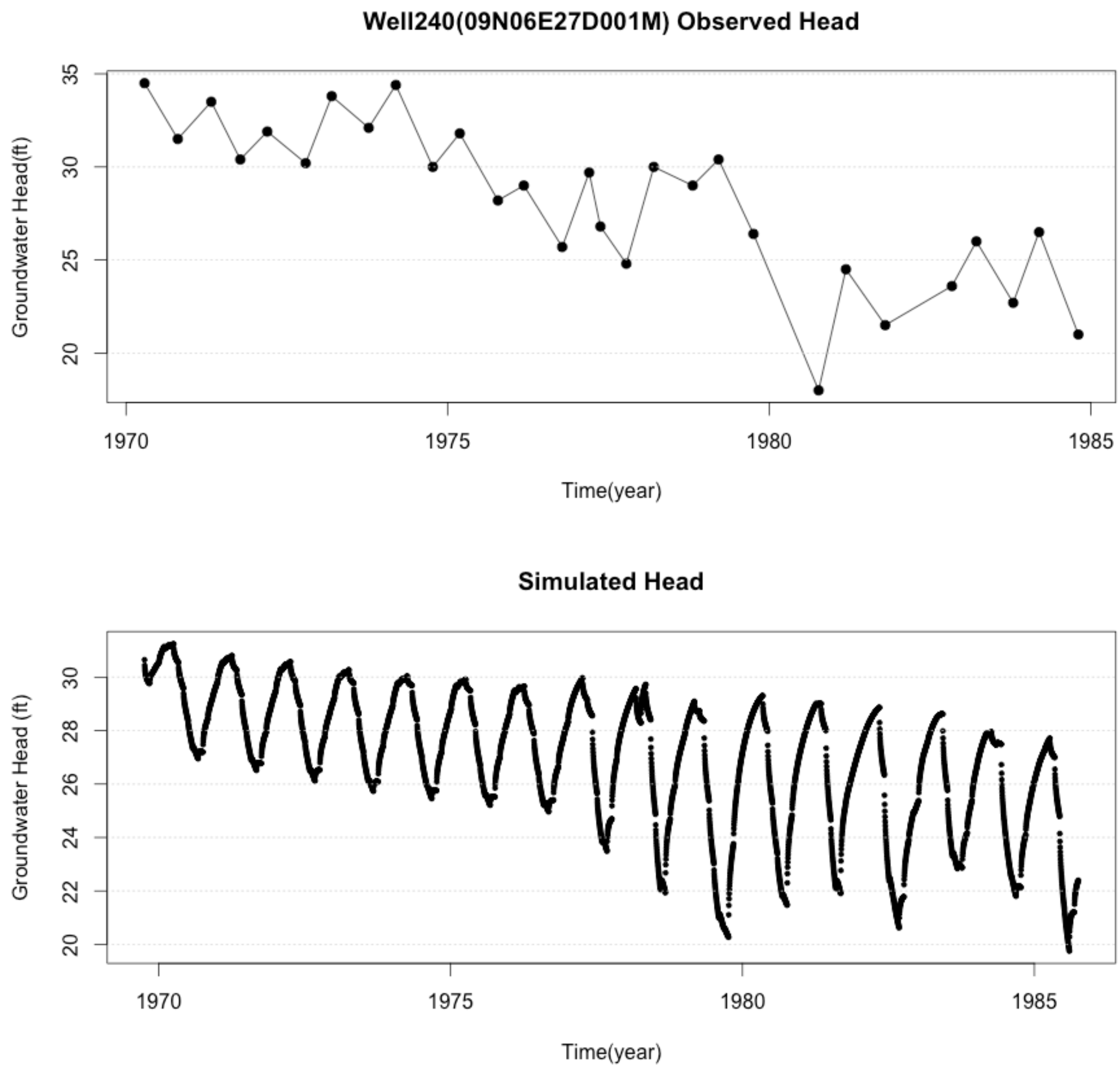
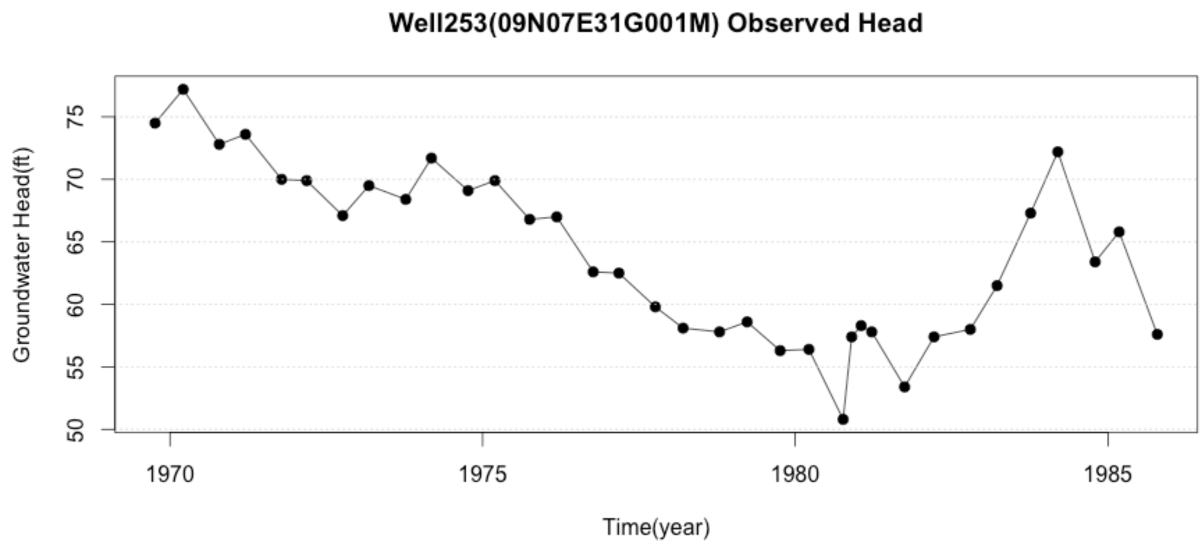


Figure D.1W comparison of simulated and observed deep confined groundwater head at well240(09N06E27D001M)



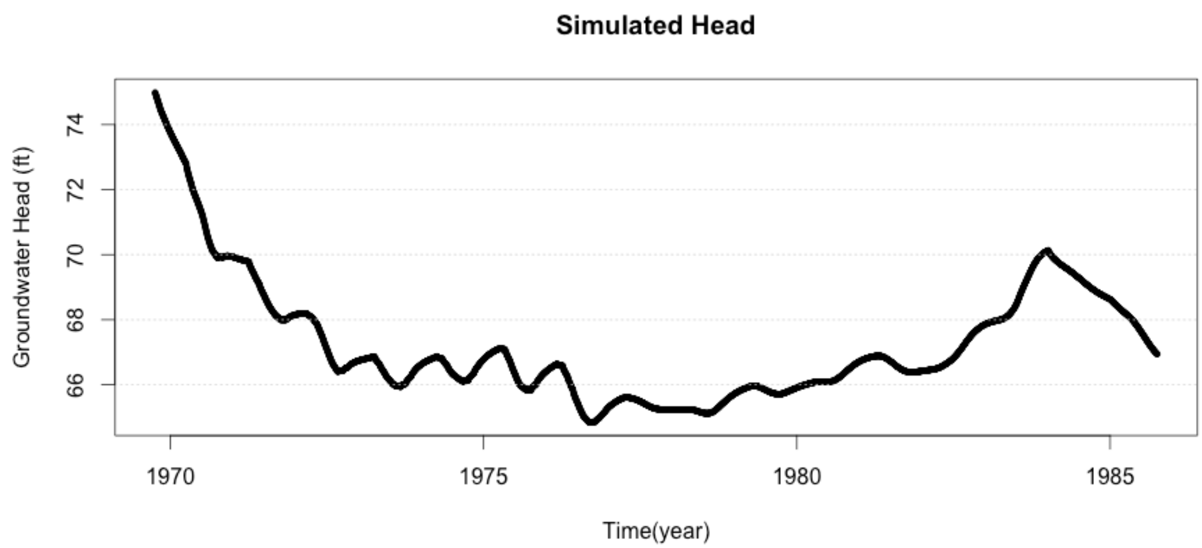
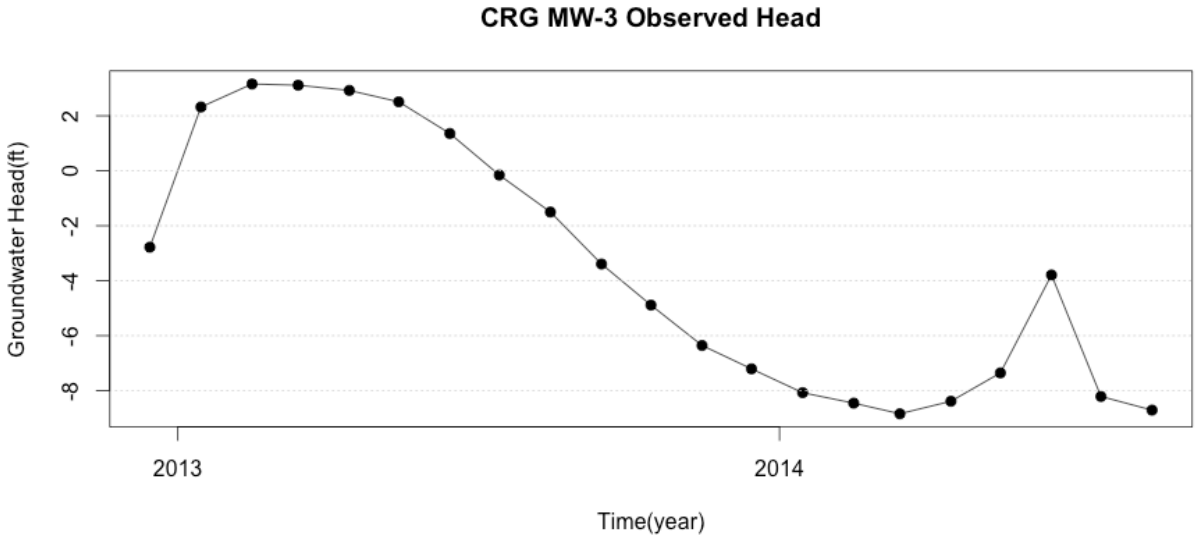


Figure D.1X comparison of simulated and observed deep confined groundwater head at well253 (09N07E31G001M)



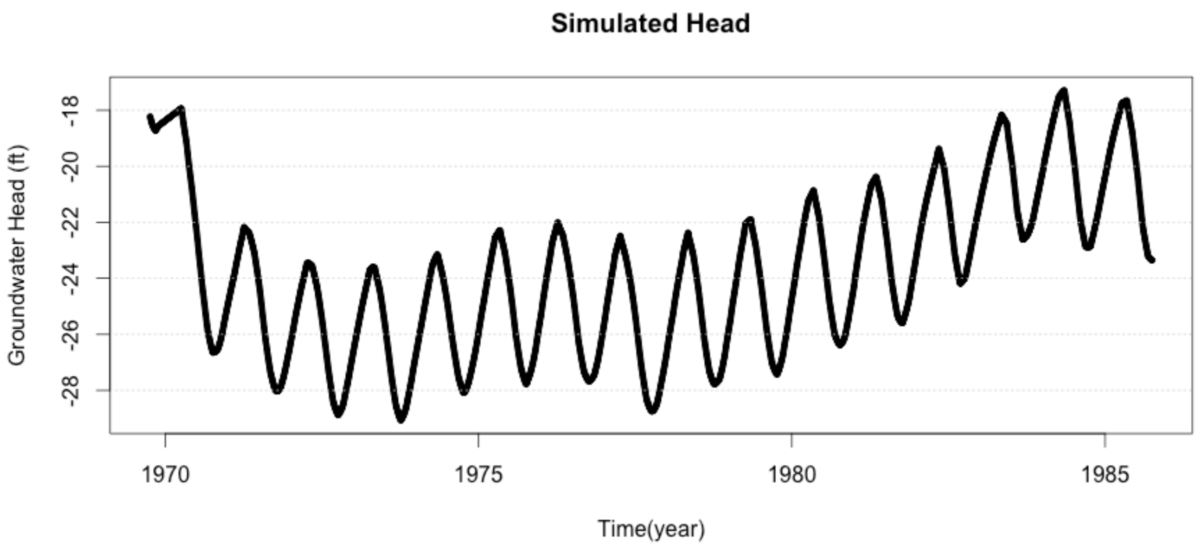


Figure D.2A comparison of simulated and observed deep confined groundwater head at CRG MW-3

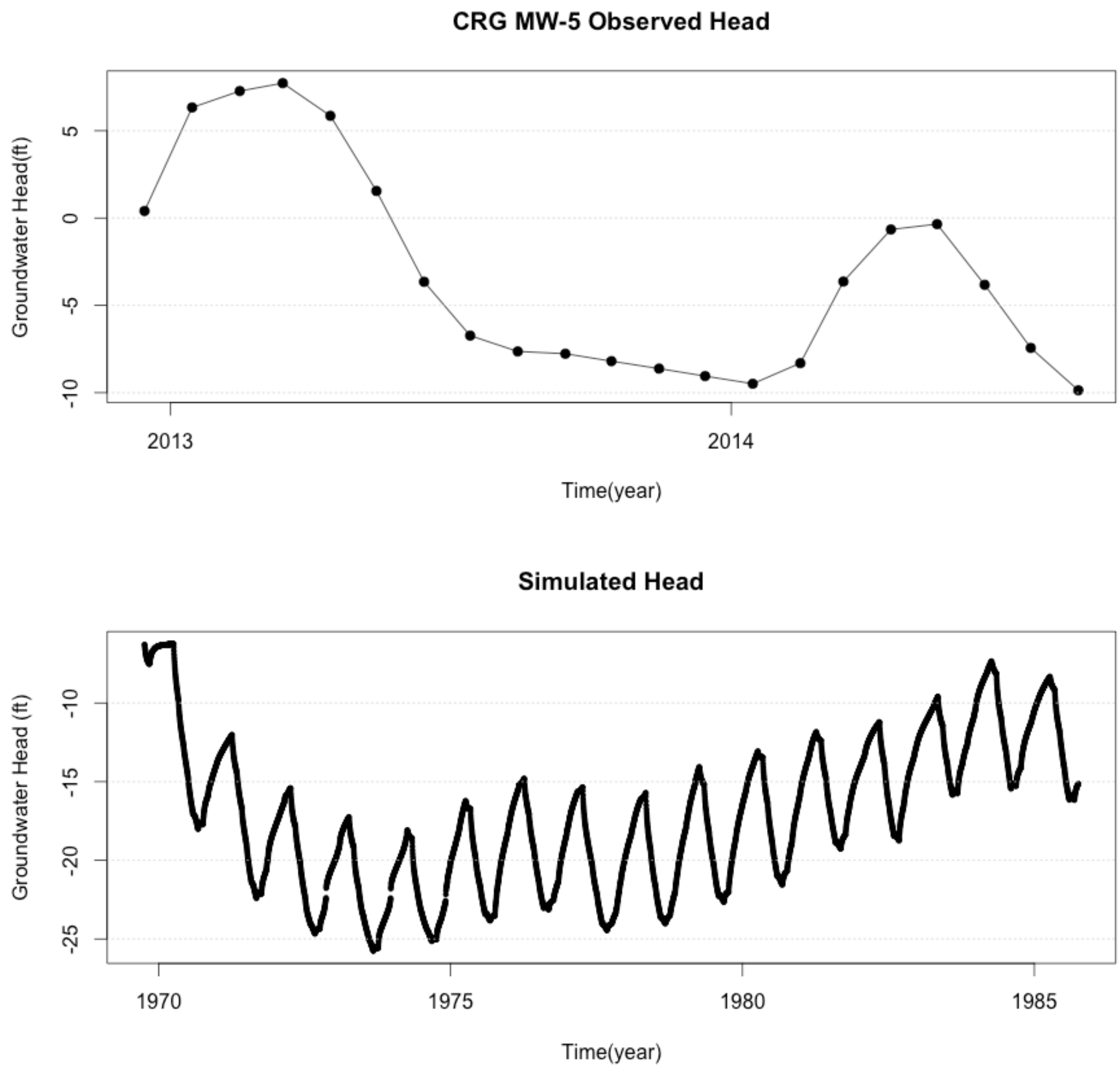


Figure D.2B comparison of simulated and observed deep confined groundwater head at CRG MW-5

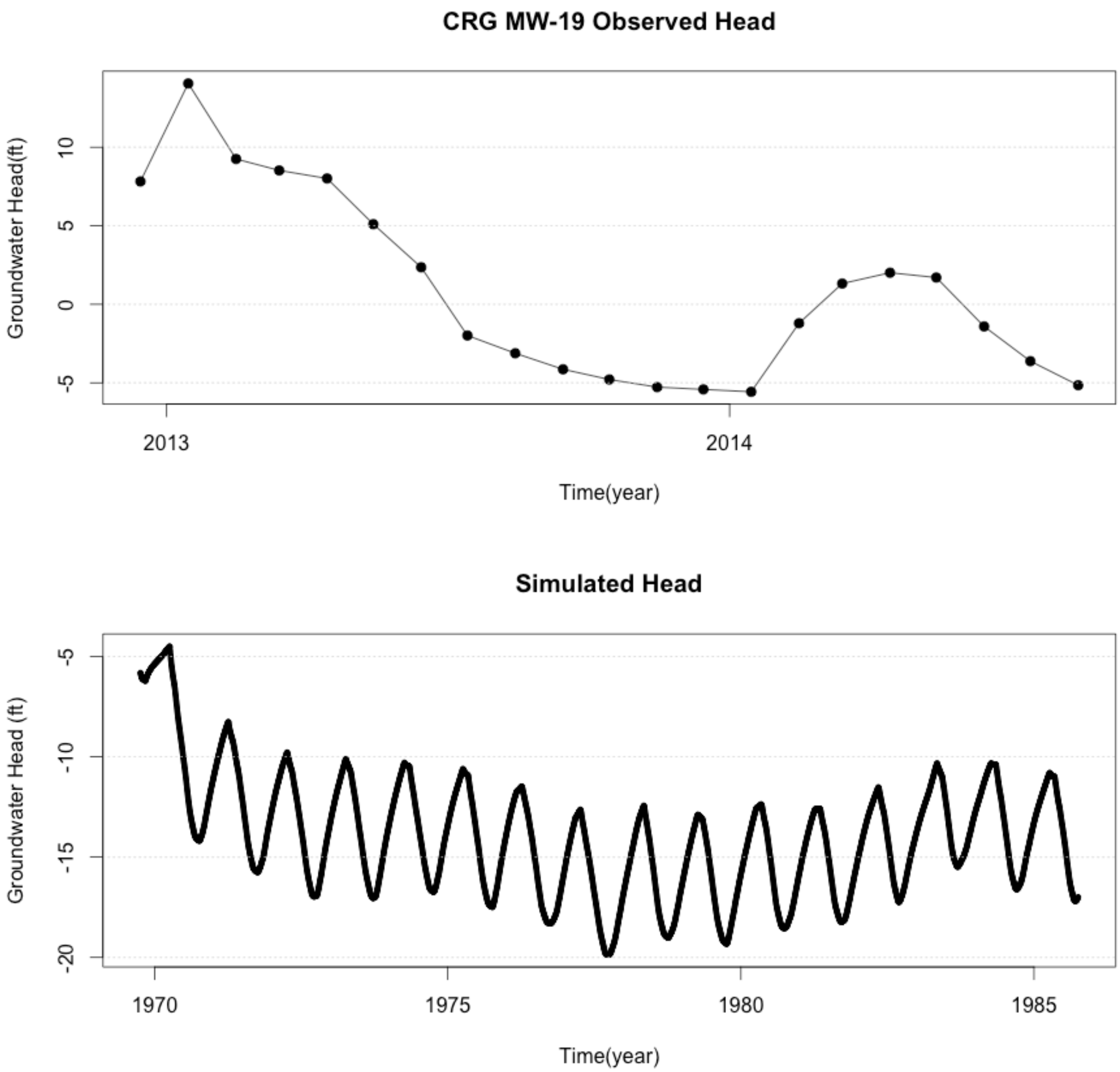


Figure D.2C comparison of simulated and observed deep confined groundwater head at CRG MW-19

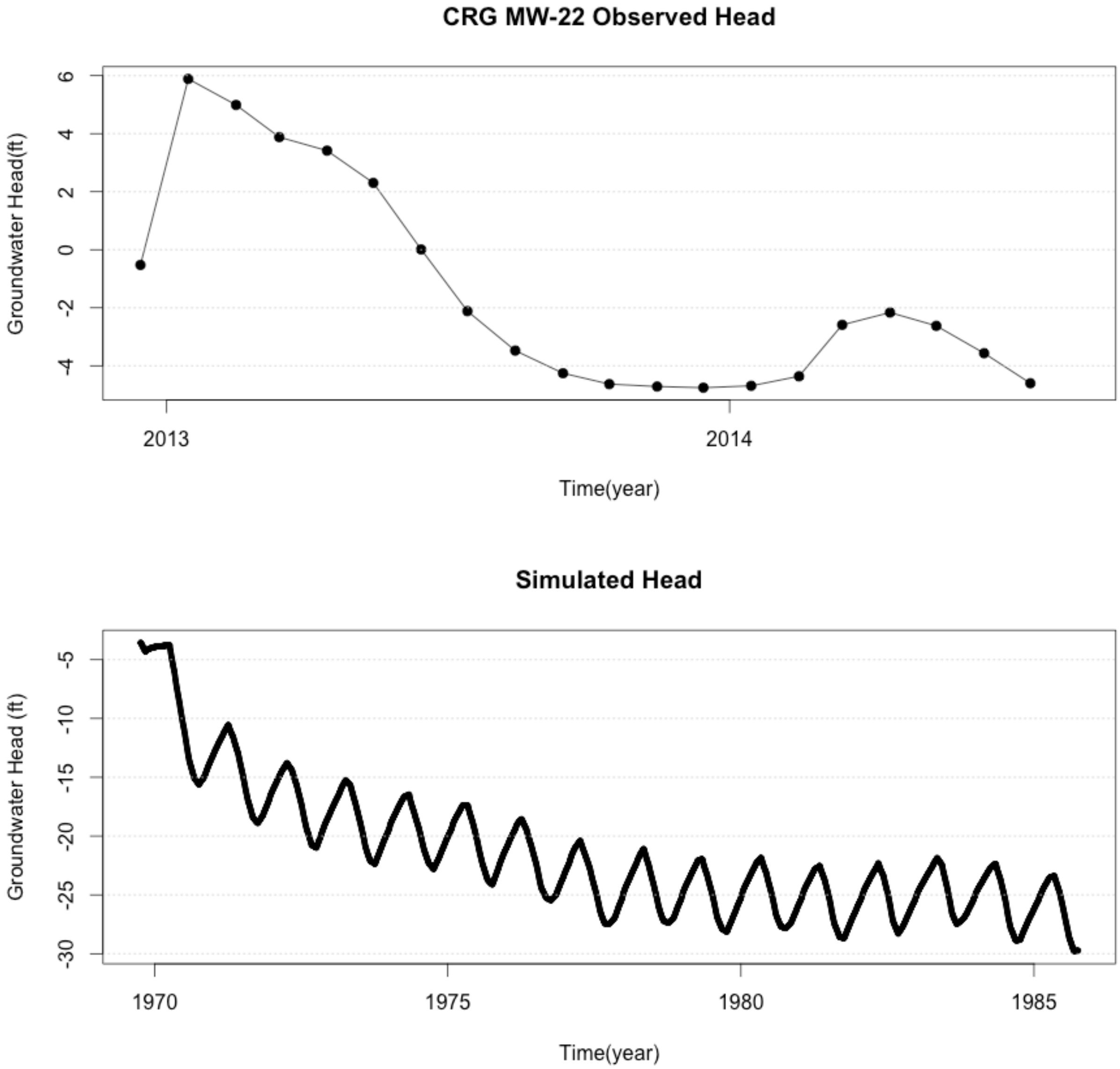
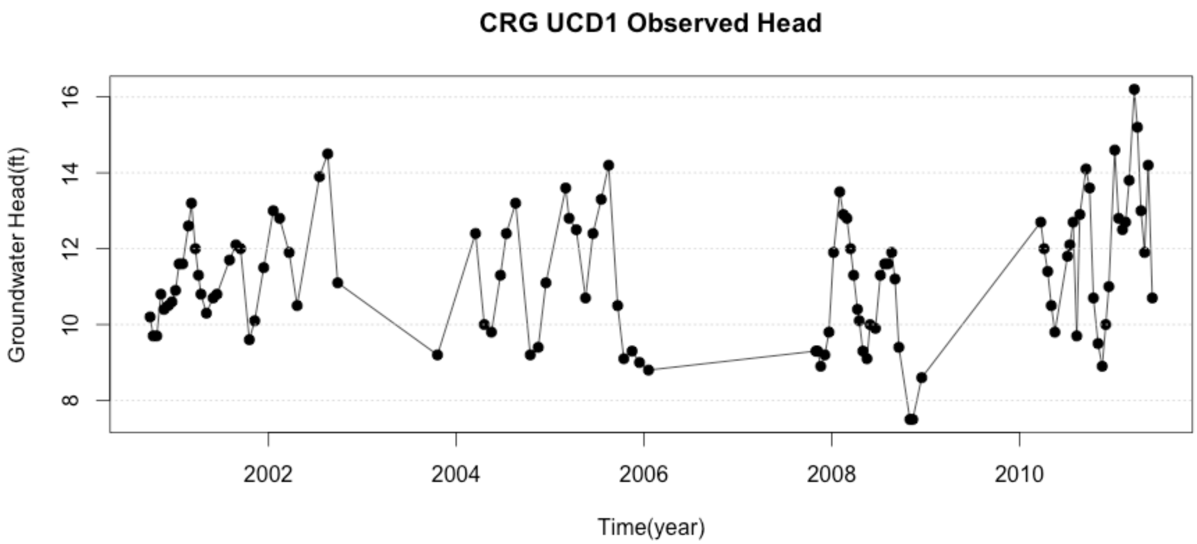


Figure D.2D comparison of simulated and observed deep confined groundwater head at CRG MW-22



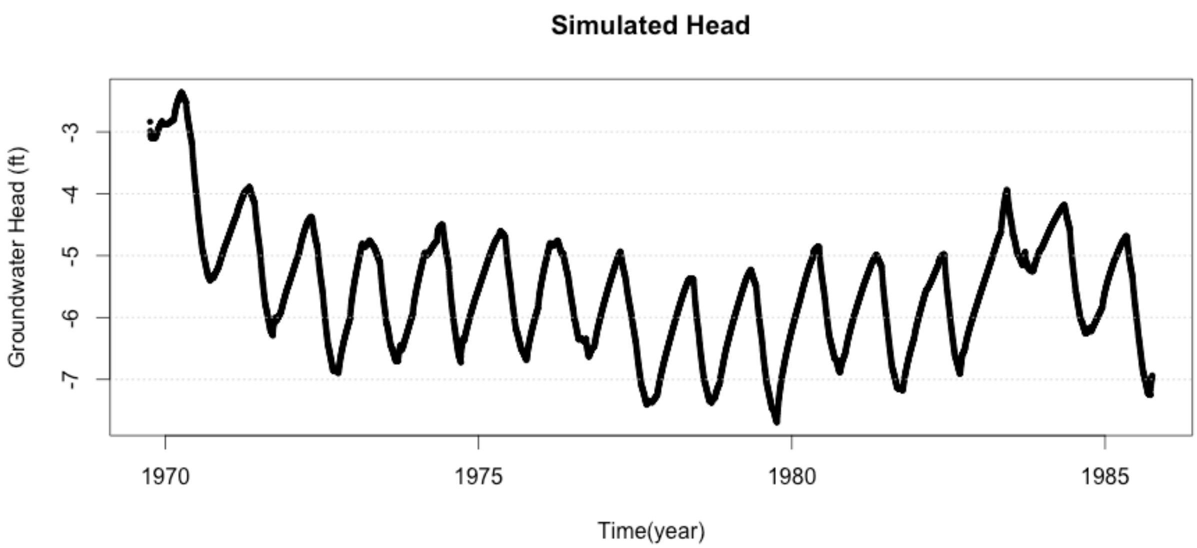


Figure D.2E comparison of simulated and observed deep confined groundwater head at CRG UCD-1

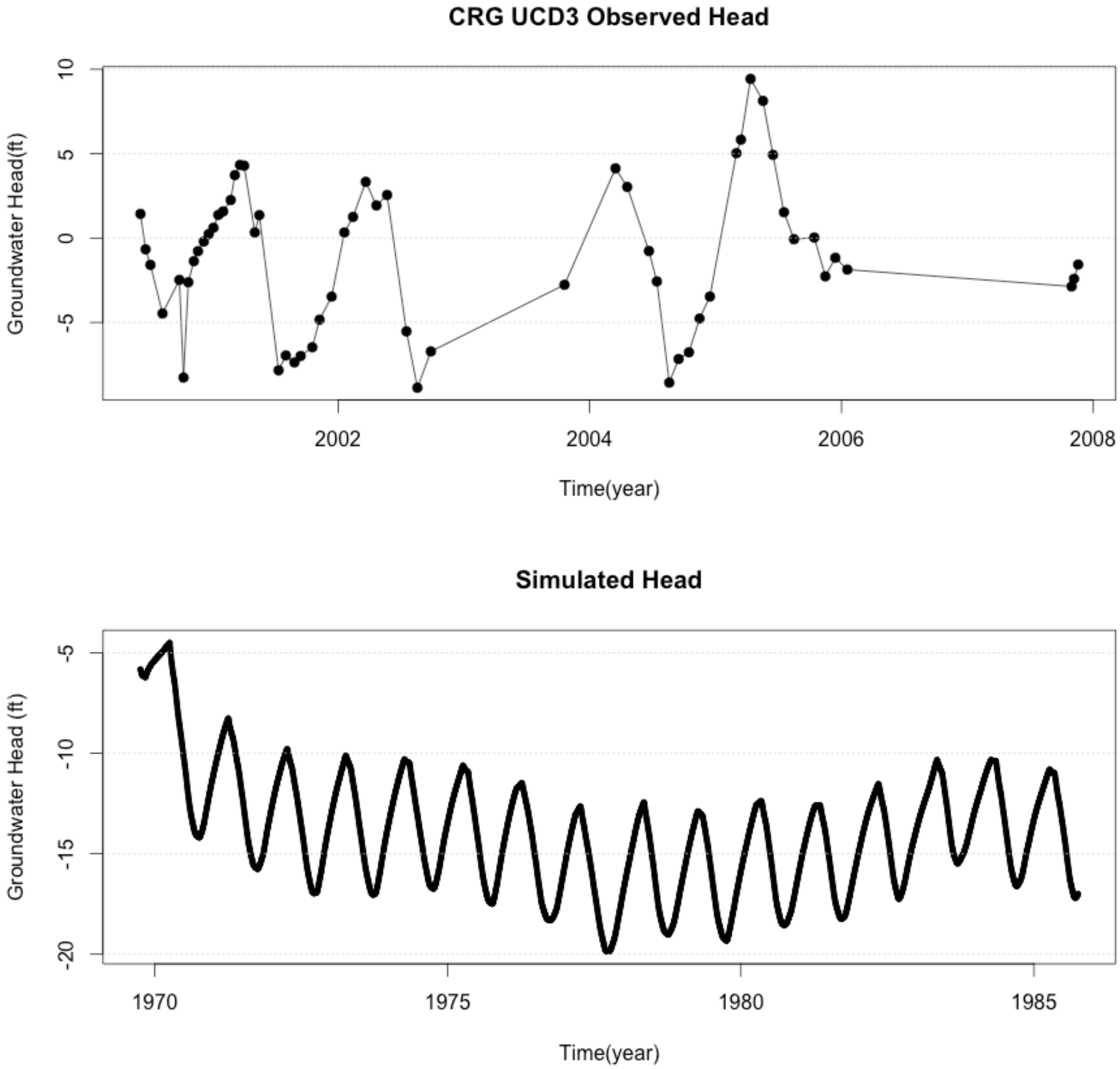


Figure D.2E comparison of simulated and observed deep confined groundwater head at CRG UCD-3

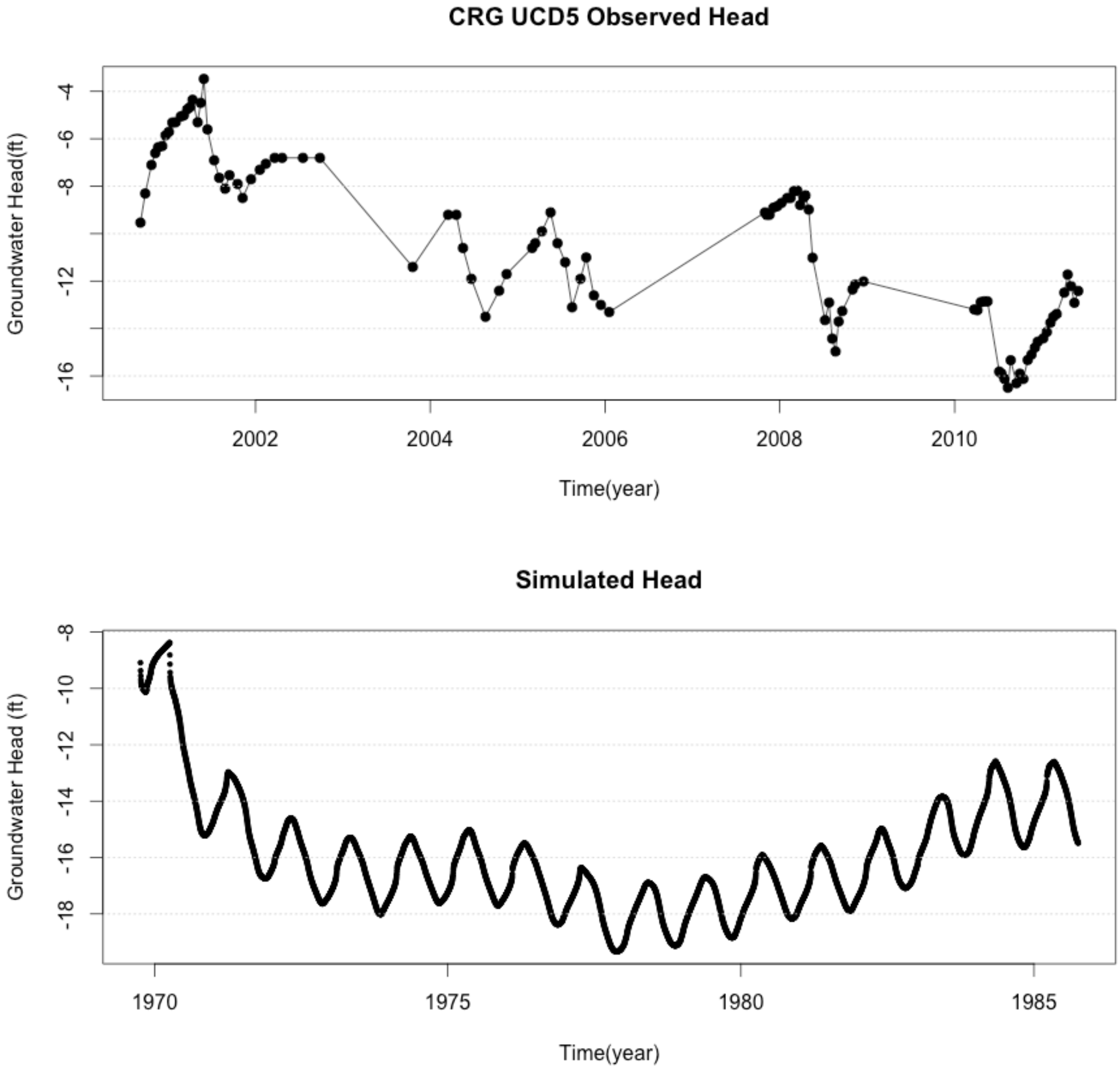


Figure D.2F comparison of simulated and observed deep confined groundwater head at CRG UCD-5

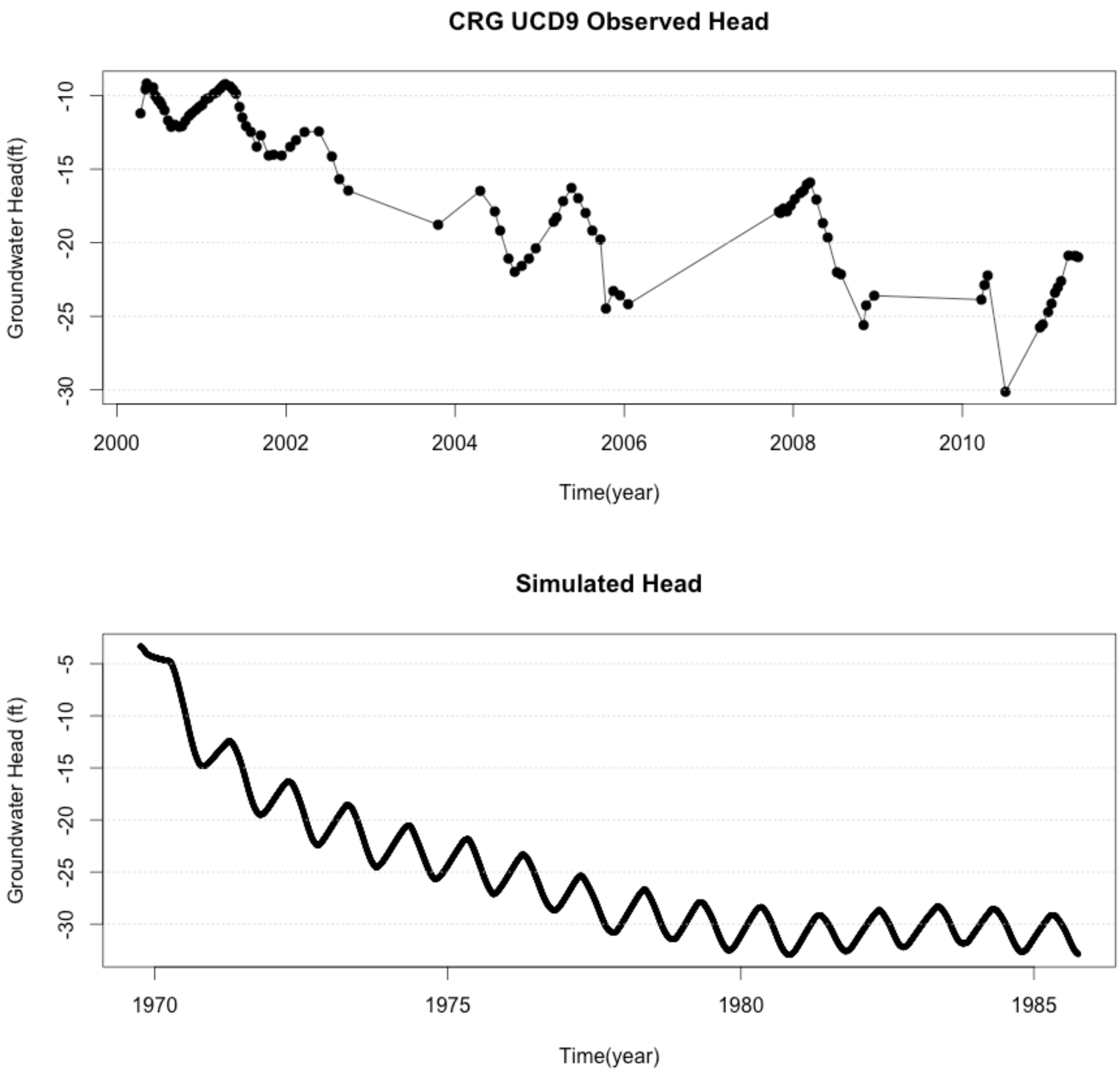


Figure D.2G comparison of simulated and observed deep confined groundwater head at CRG UCD-9

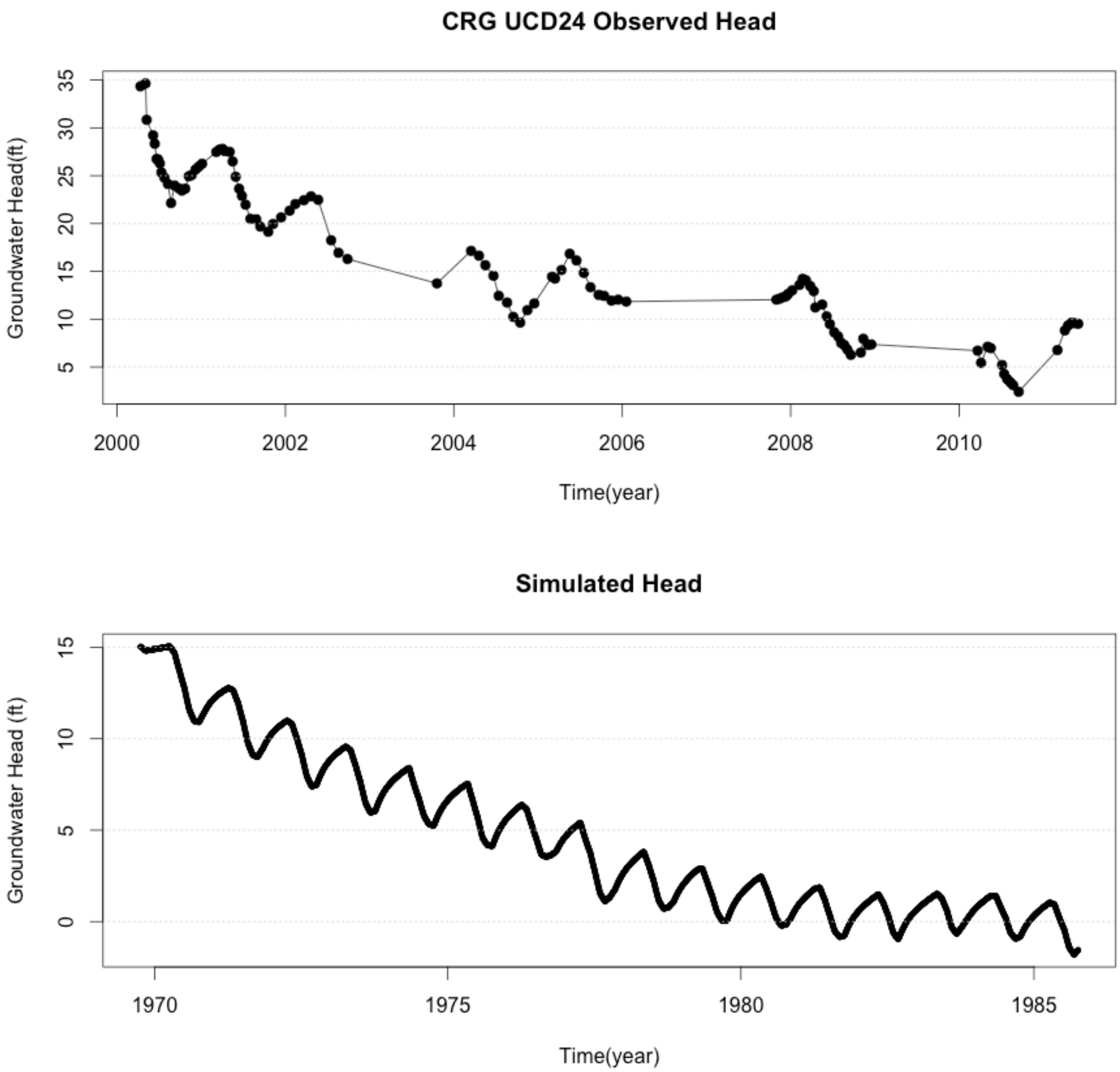
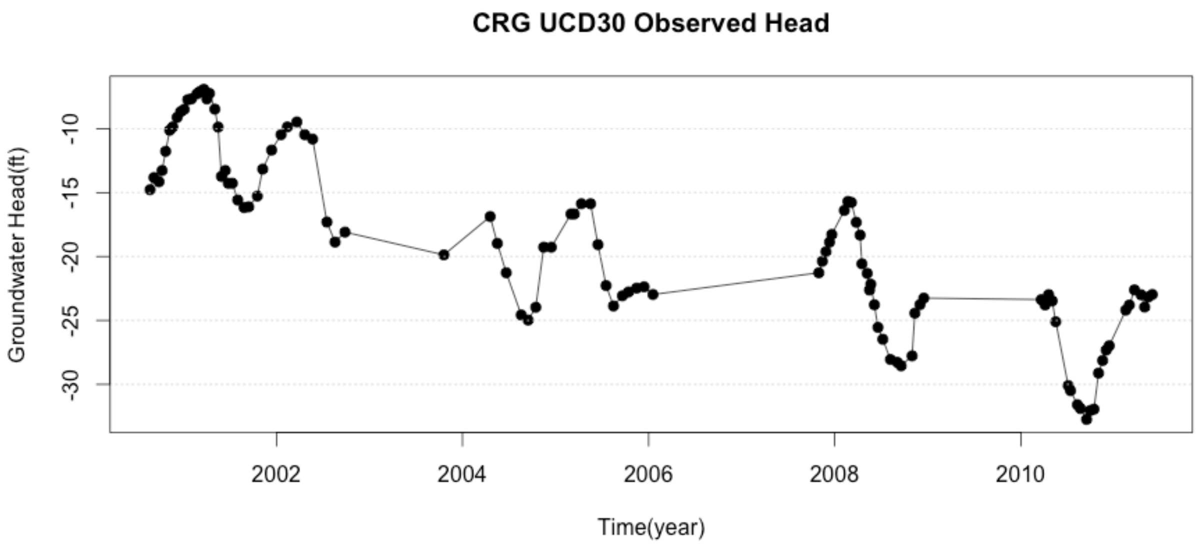


Figure D.2H comparison of simulated and observed deep confined groundwater head at CRG UCD-24



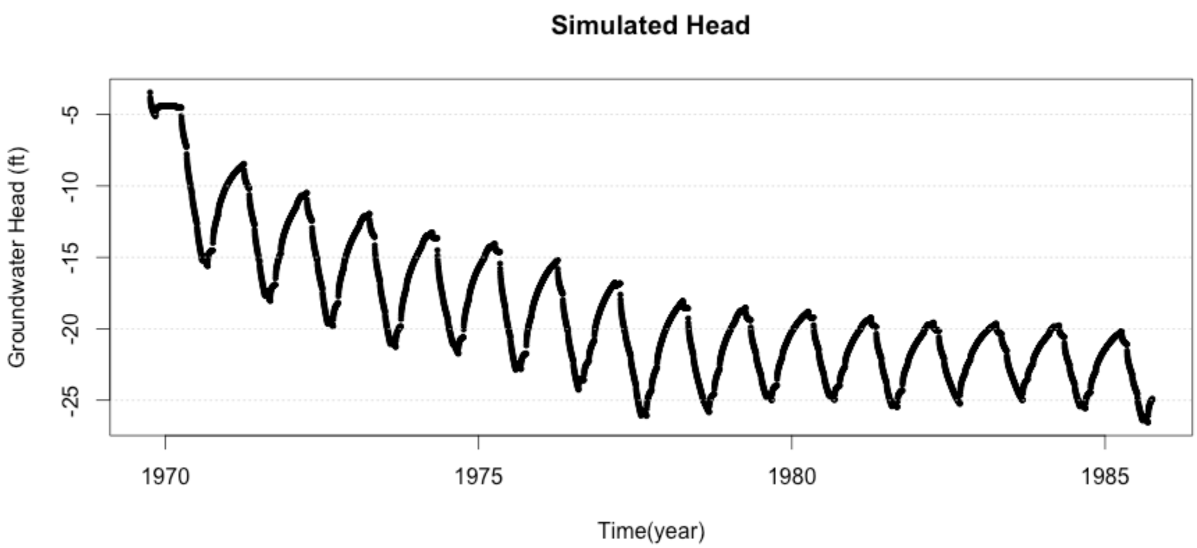


Figure D.2I comparison of simulated and observed deep confined groundwater head at CRG UCD-30

D.2 Calibration-local performance

The second calibration goal is to examine whether the simulated groundwater head matches with the observed groundwater head at selected DWR monitoring wells. 39 monitoring wells among 200 wells that fall within the modeling domain is selected (Figure D.3). Comparisons between simulated groundwater head and observed groundwater head at selected calibration wells are shown in Figure D.4.

For all of the selected calibration wells, the simulated groundwater head pattern matches pretty well with the observed groundwater head pattern. Remember that perfect match is not likely to be achieved, because the heterogeneous geological model represents only one geological structure among many possible structures that honors field geological data.

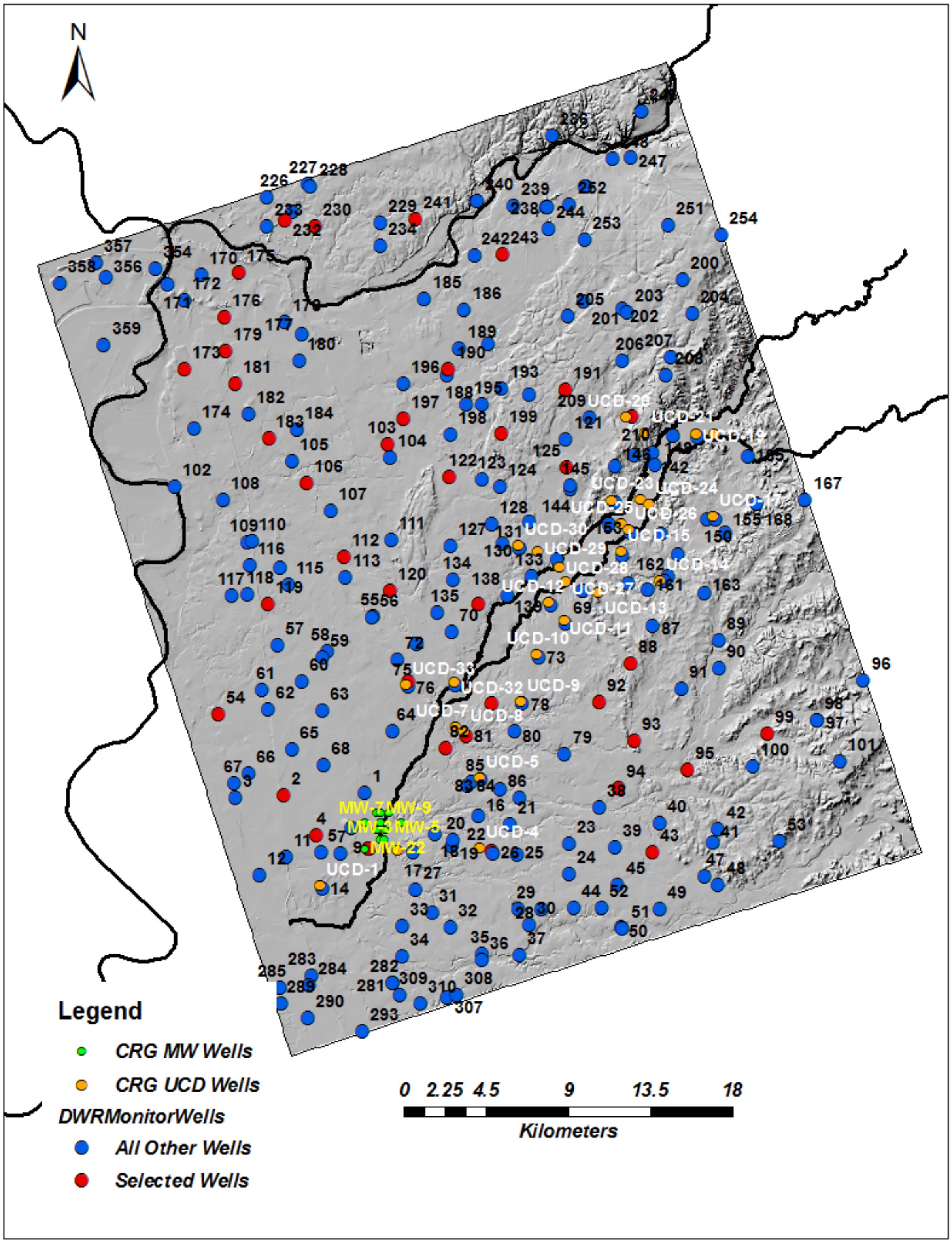
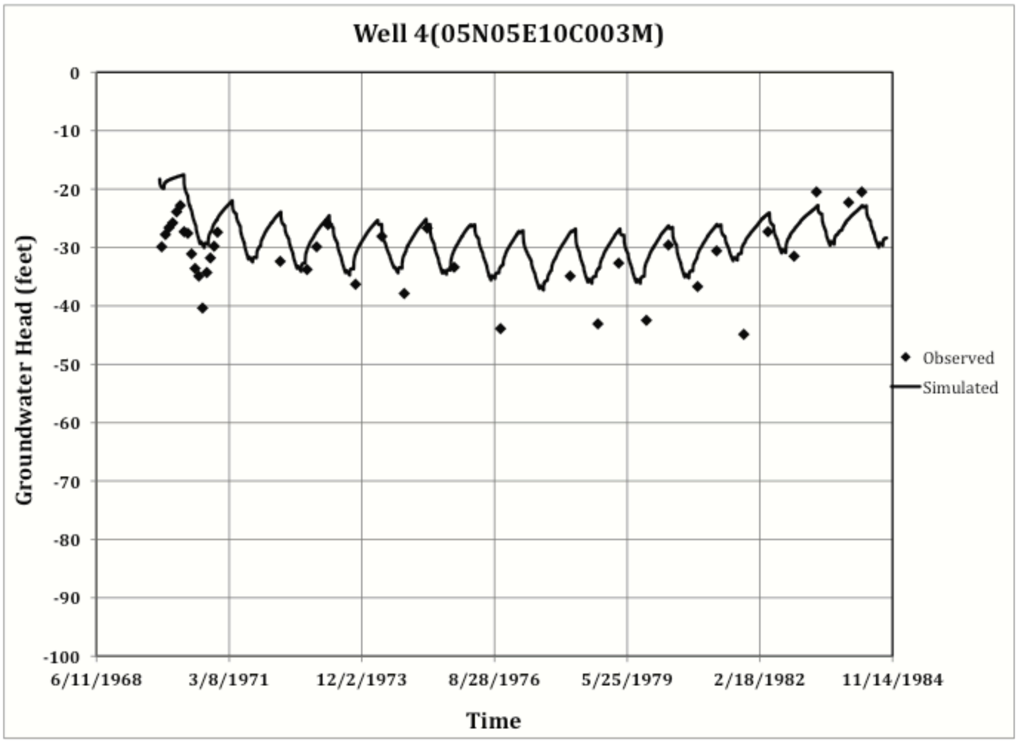
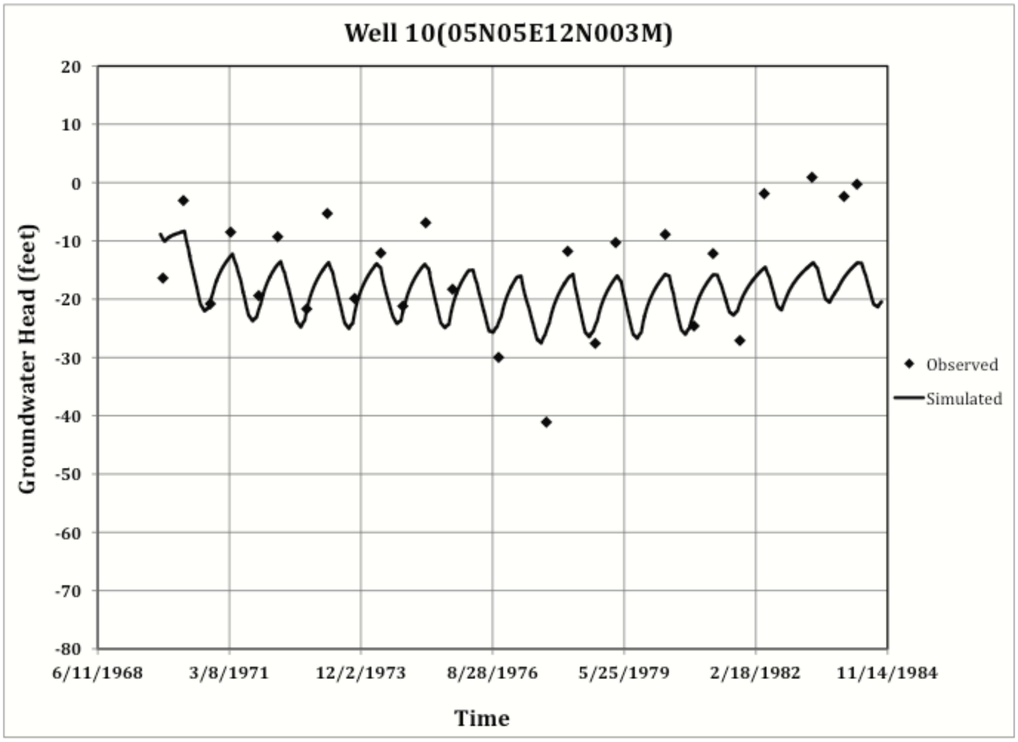
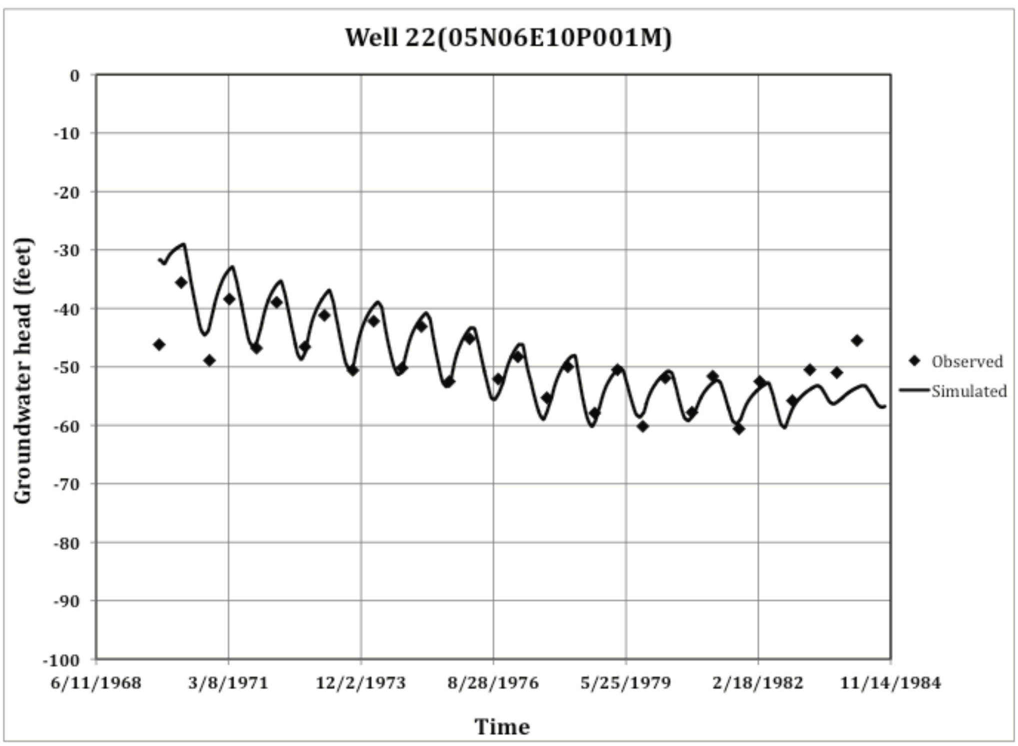
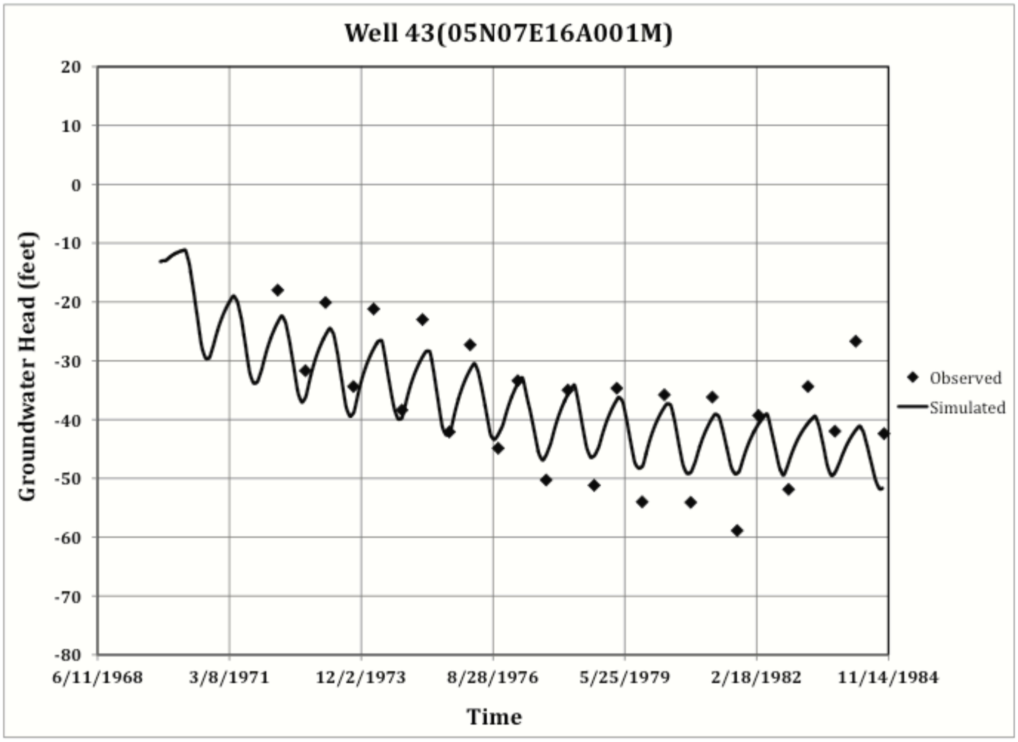

N
Legend
CRG MW Wells
CRG UCD Wells
DWRMonitorWells
All Other Wells
Selected Wells
0 2.25 4.5 9 13.5 18
Kilometers

Figure D.3 all of the DWR monitor wells in model domain and selected representative calibration well

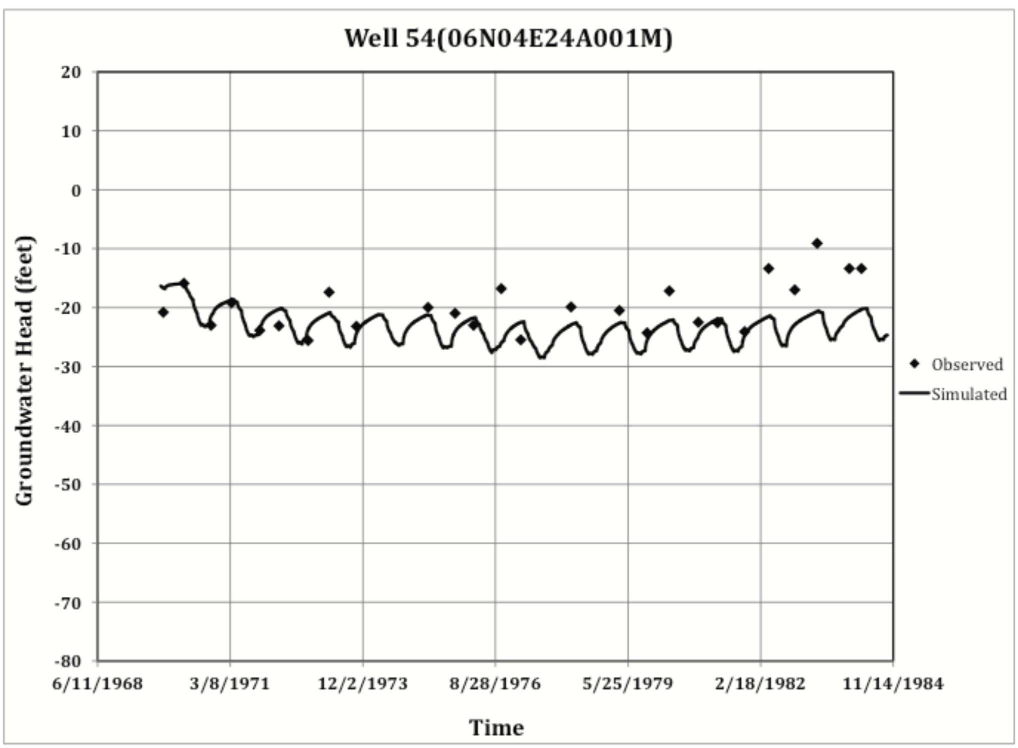



Well 10(05N05E12N003M)
Groundwater Head (feet)
20
10
0
-10
-20
-30
-40
-50
-60
-70
-80
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated

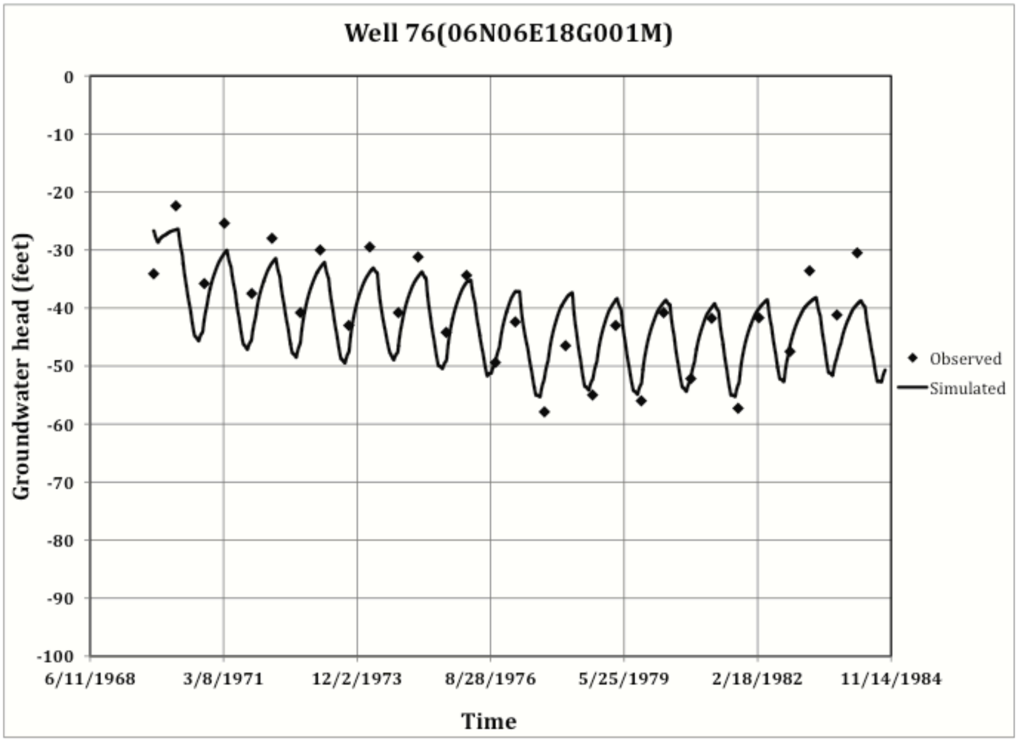

Well 22(05N06E10P001M)
Groundwater head (feet)
0
-10
-20
-30
-40
-50
-60
-70
-80
-90
-100
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated



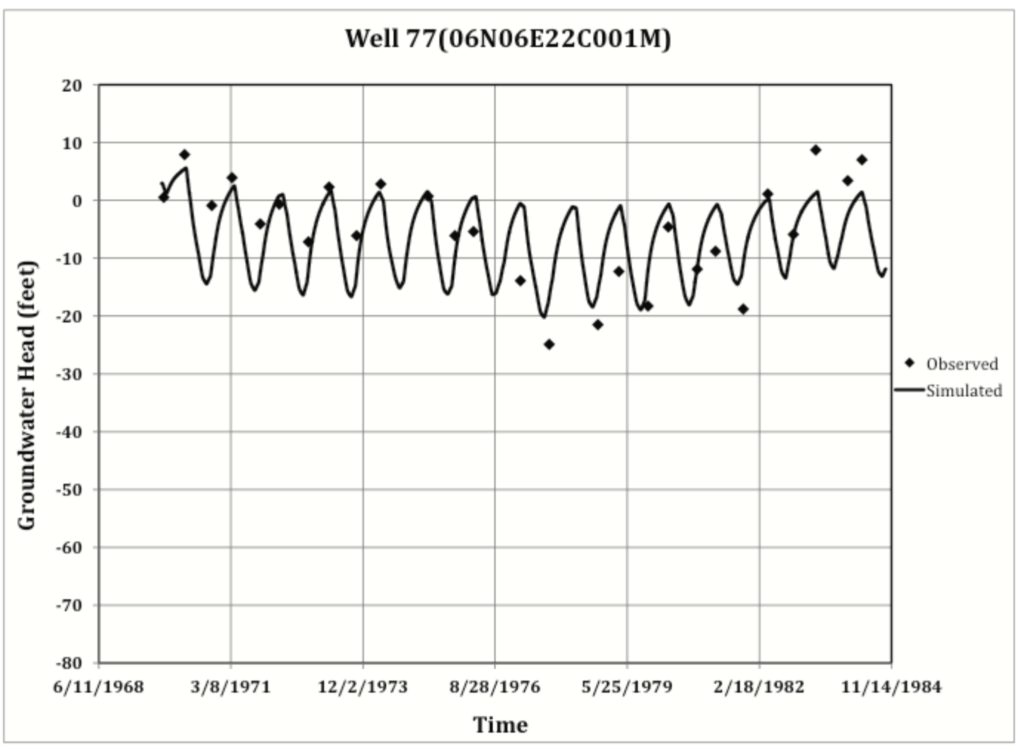
Well 43(05N07E16A001M)
Groundwater Head (feet)
20
10
0
-10
-20
-30
-40
-50
-60
-70
-80
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated



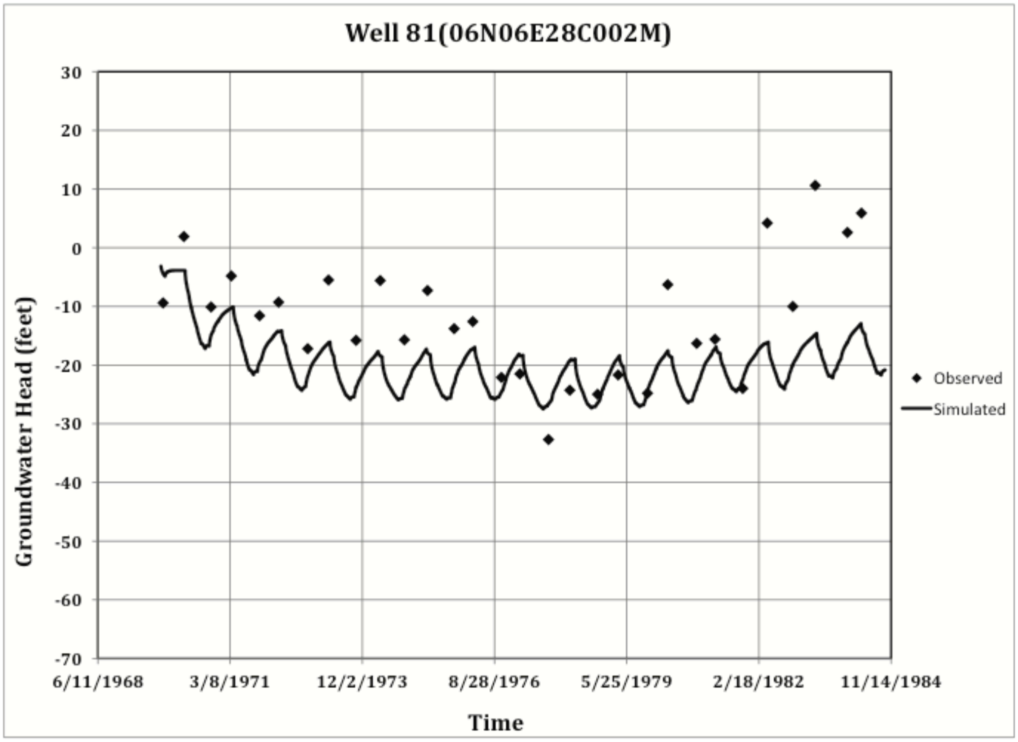
Well 54(06N04E24A001M)
Groundwater Head (feet)
20
10
0
-10
-20
-30
-40
-50
-60
-70
-80
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated

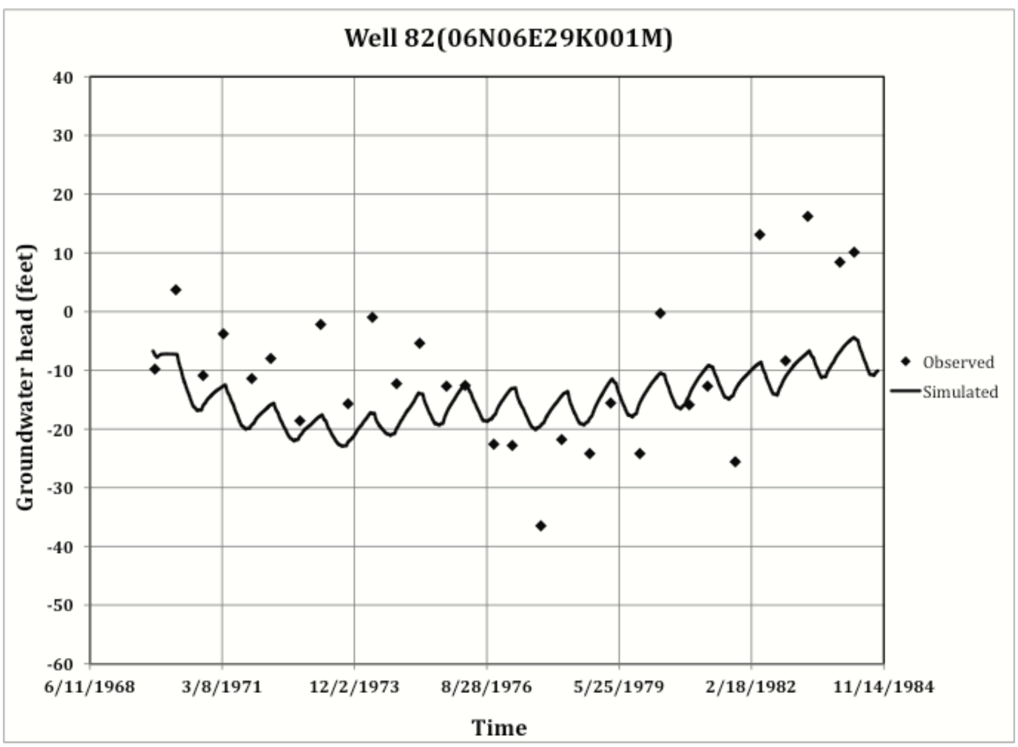


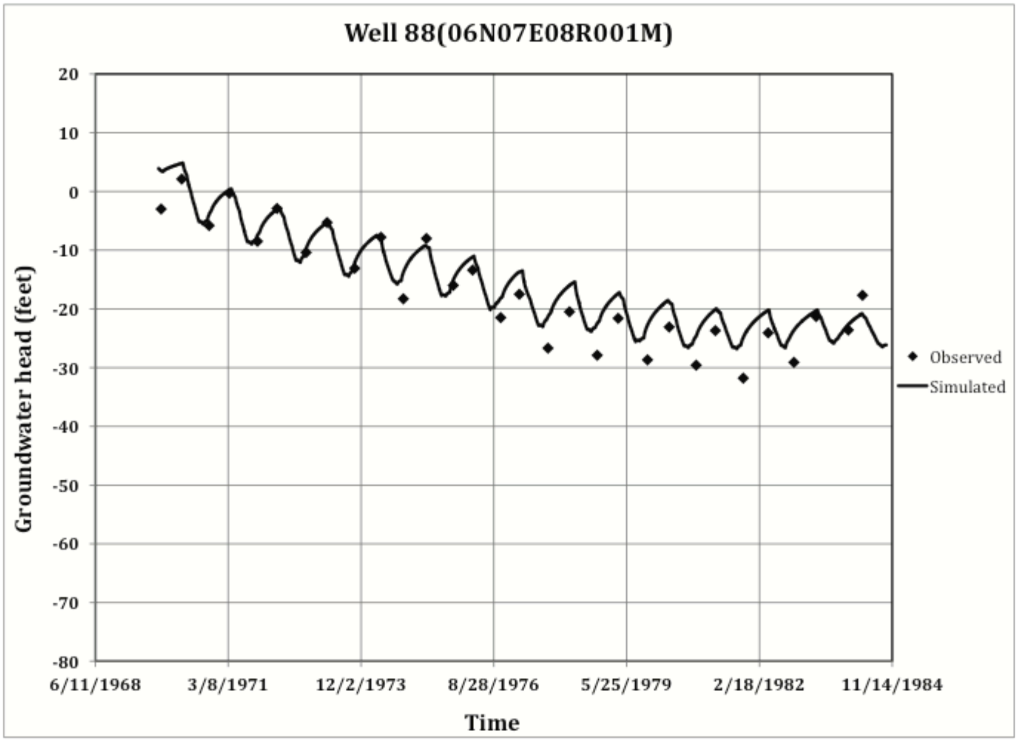
Well 76(06N06E18G001M)
Groundwater head (feet)
0
-10
-20
-30
-40
-50
-60
-70
-80
-90
-100
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated

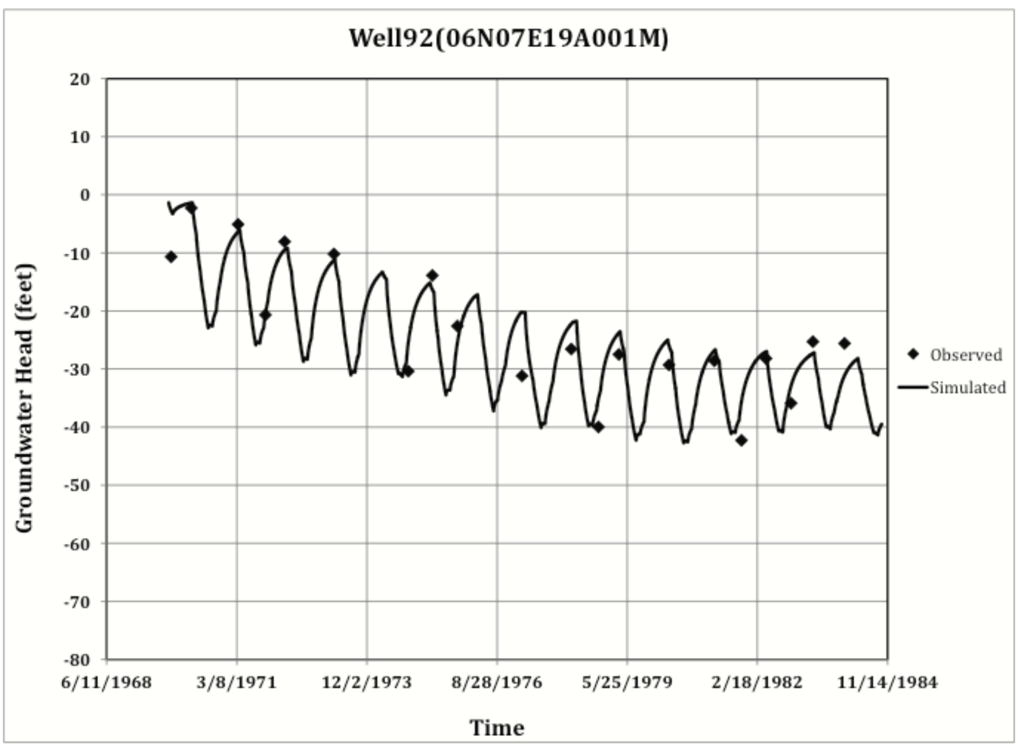


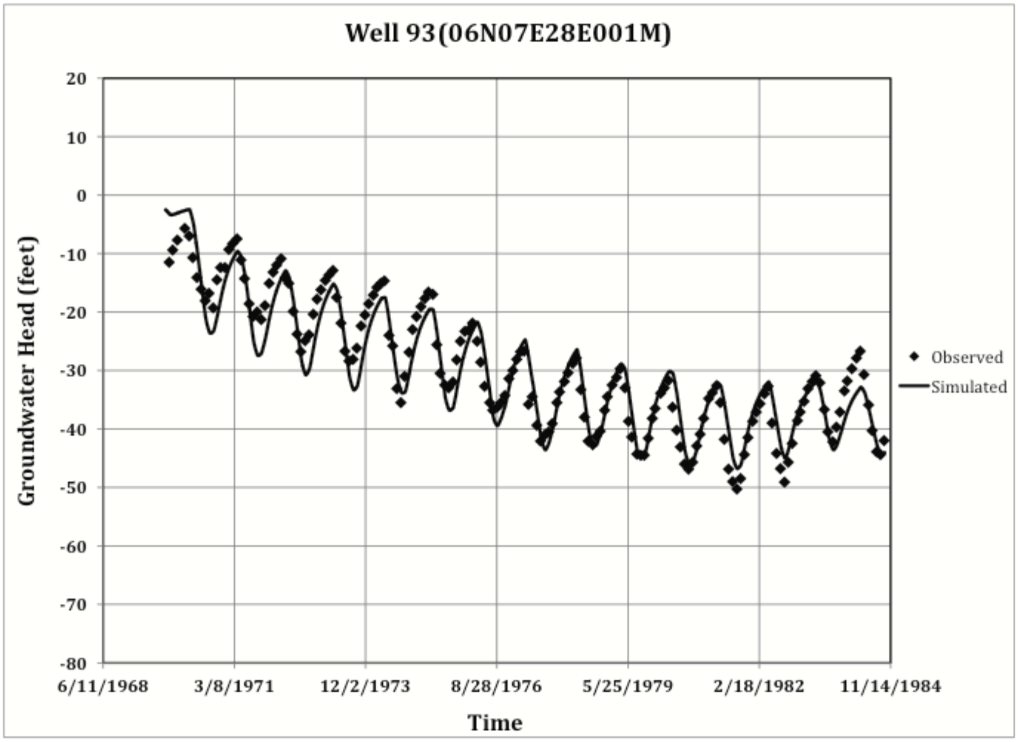
Well 77(06N06E22C001M)
Groundwater Head (feet)
20
10
0
-10
-20
-30
-40
-50
-60
-70
-80
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated

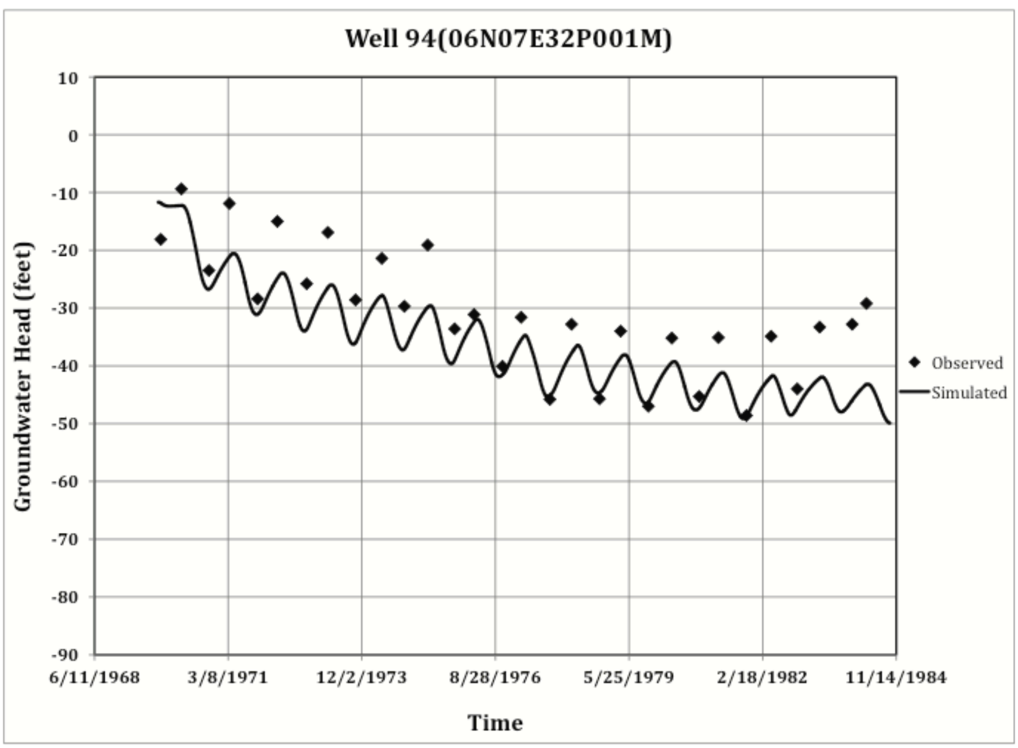

Well 81(06N06E28C002M)
Groundwater Head (feet)
30
20
10
0
-10
-20
-30
-40
-50
-60
-70
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated

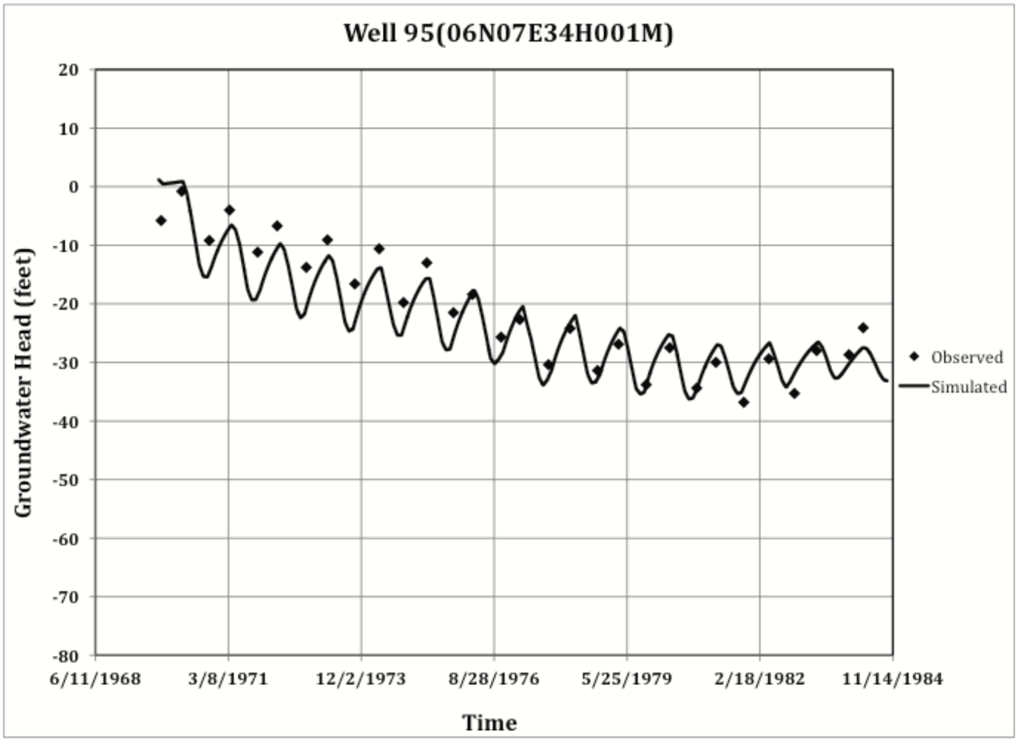

Well 82(06N06E29K001M)
Groundwater head (feet)
40
30
20
10
0
-10
-20
-30
-40
-50
-60
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated


Well 88(06N07E08R001M)
Groundwater head (feet)
20
10
0
-10
-20
-30
-40
-50
-60
-70
-80
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated

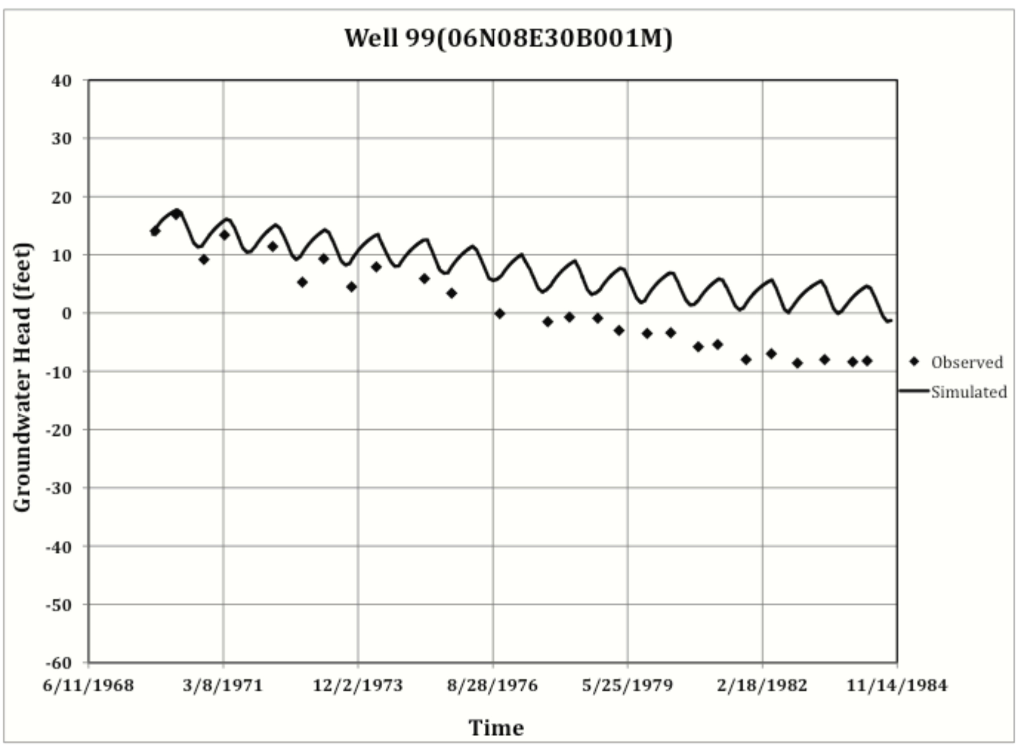

Well92(06N07E19A001M)
Groundwater Head (feet)
20
10
0
-10
-20
-30
-40
-50
-60
-70
-80
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated


Well 93(06N07E28E001M)
Groundwater Head (feet)
20
10
0
-10
-20
-30
-40
-50
-60
-70
-80
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated

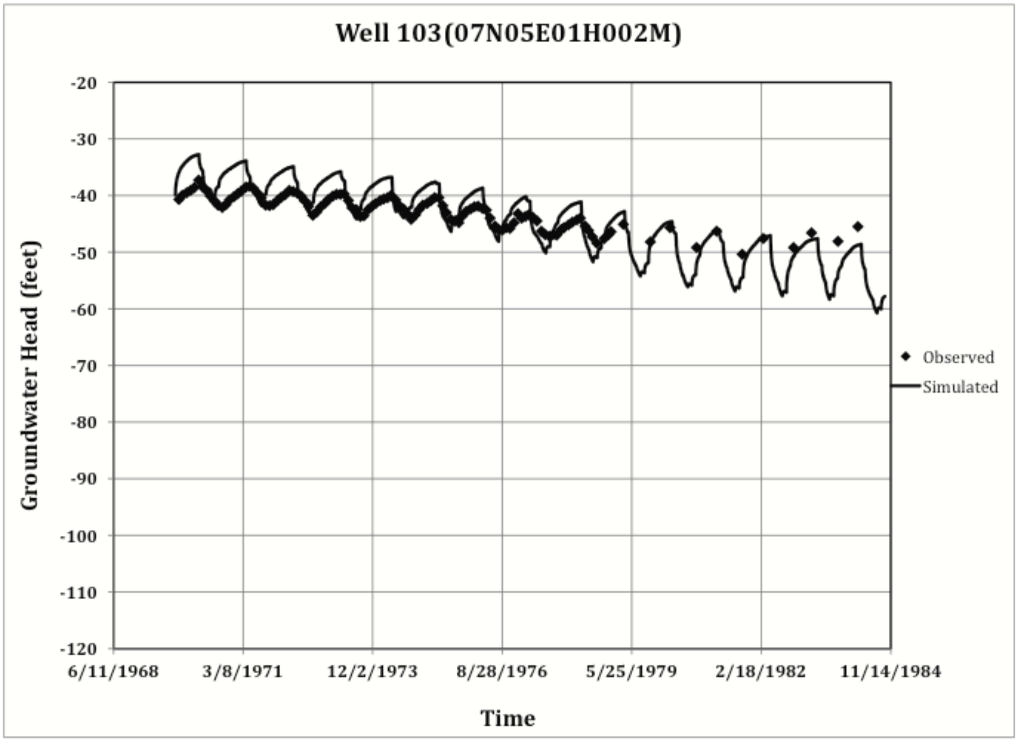

Well 94(06N07E32P001M)
Groundwater Head (feet)
10
0
-10
-20
-30
-40
-50
-60
-70
-80
-90
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated

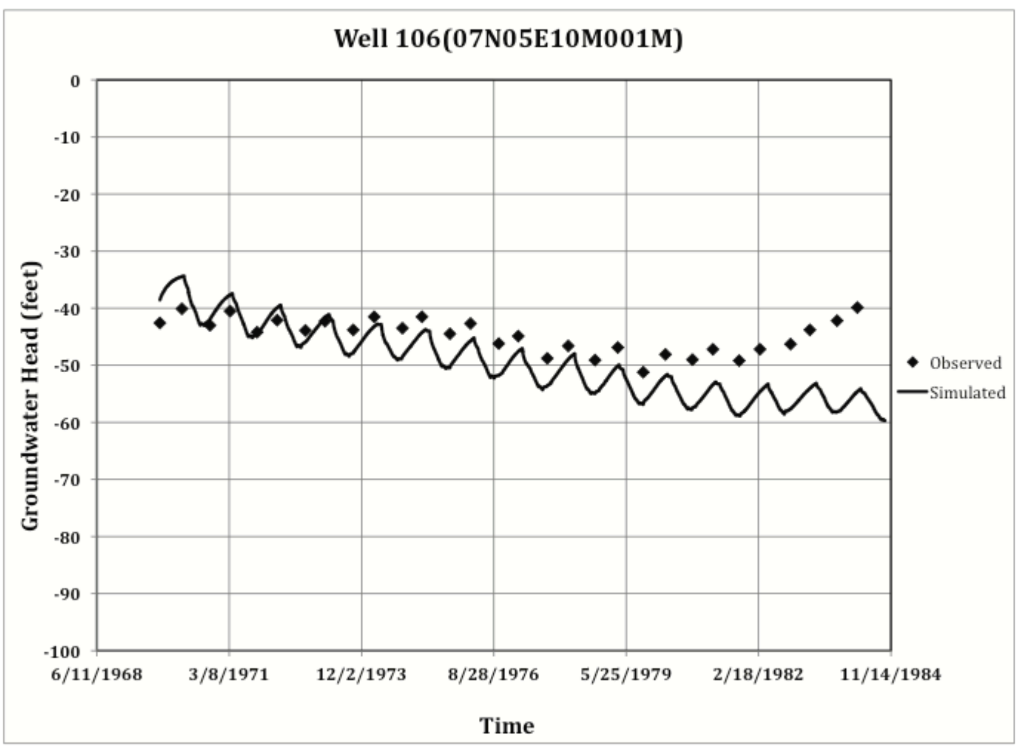


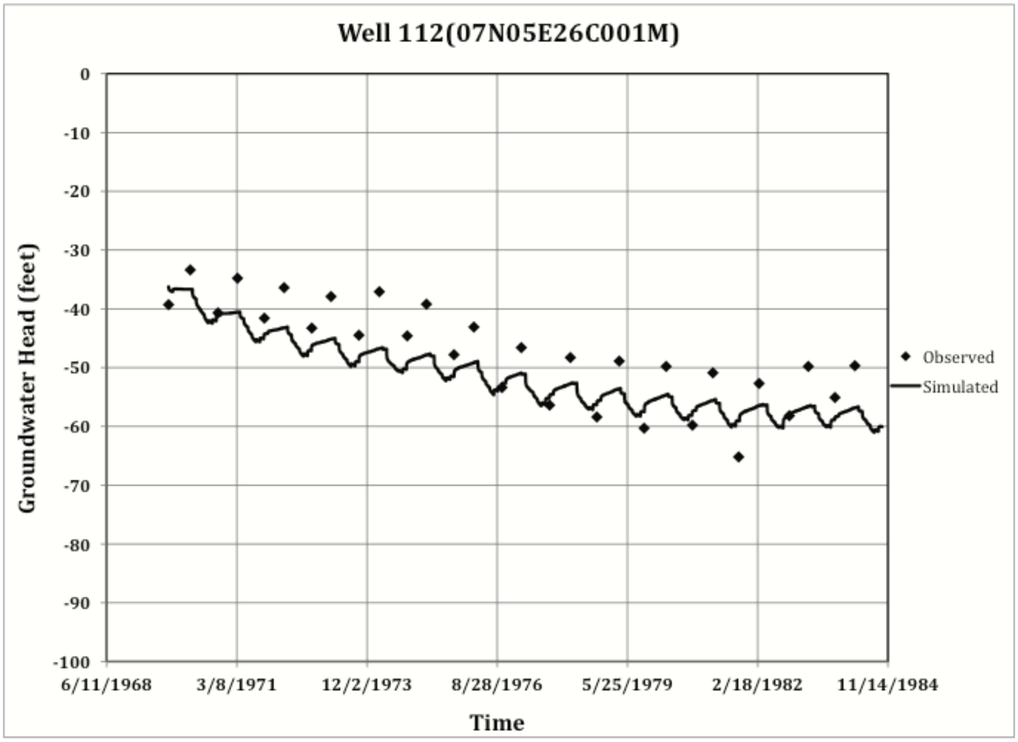
Well 95(06N07E34H001M)
20
10
0
-10
-20
-30
-40
-50
-60
-70
-80
Groundwater Head (feet)
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated

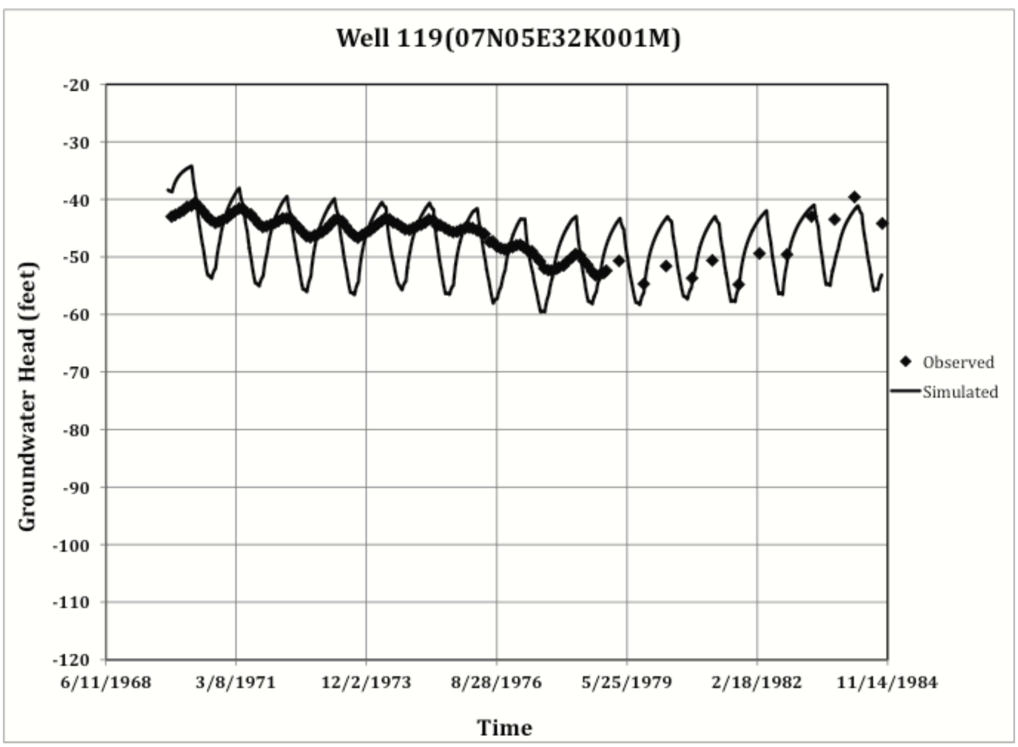


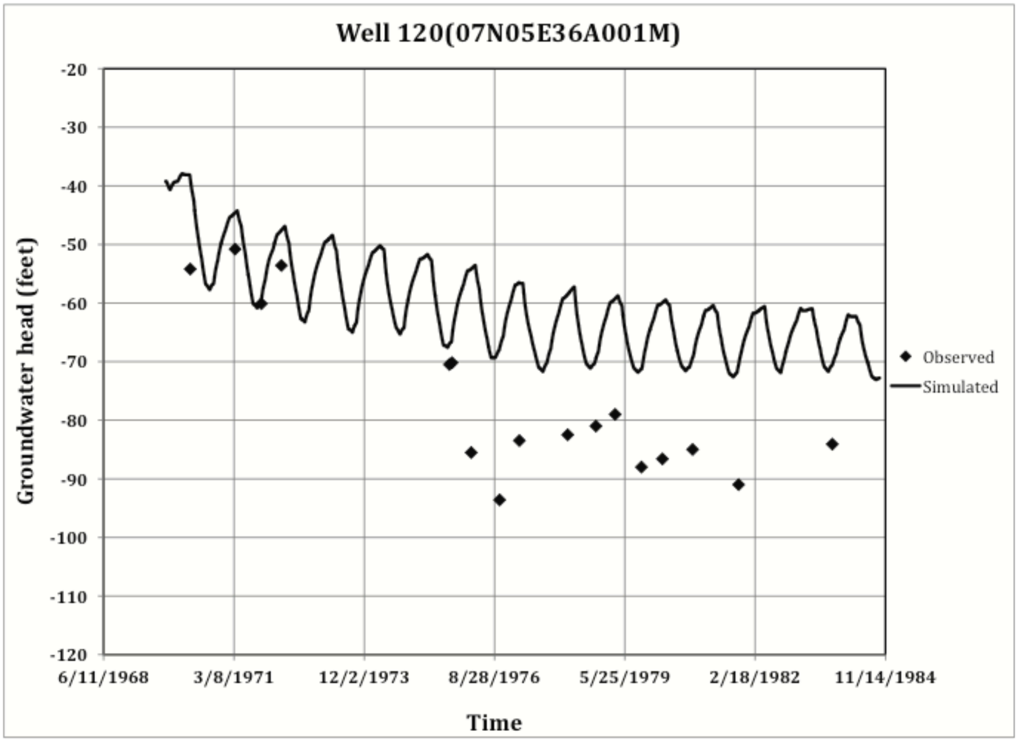
Well 99(06N08E30B001M)
40
30
20
10
0
-10
-20
-30
-40
-50
-60
Groundwater Head (feet)
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated


Well 103(07N05E01H002M)
-20
-30
-40
-50
-60
-70
-80
-90
-100
-110
-120
Groundwater Head (feet)
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated

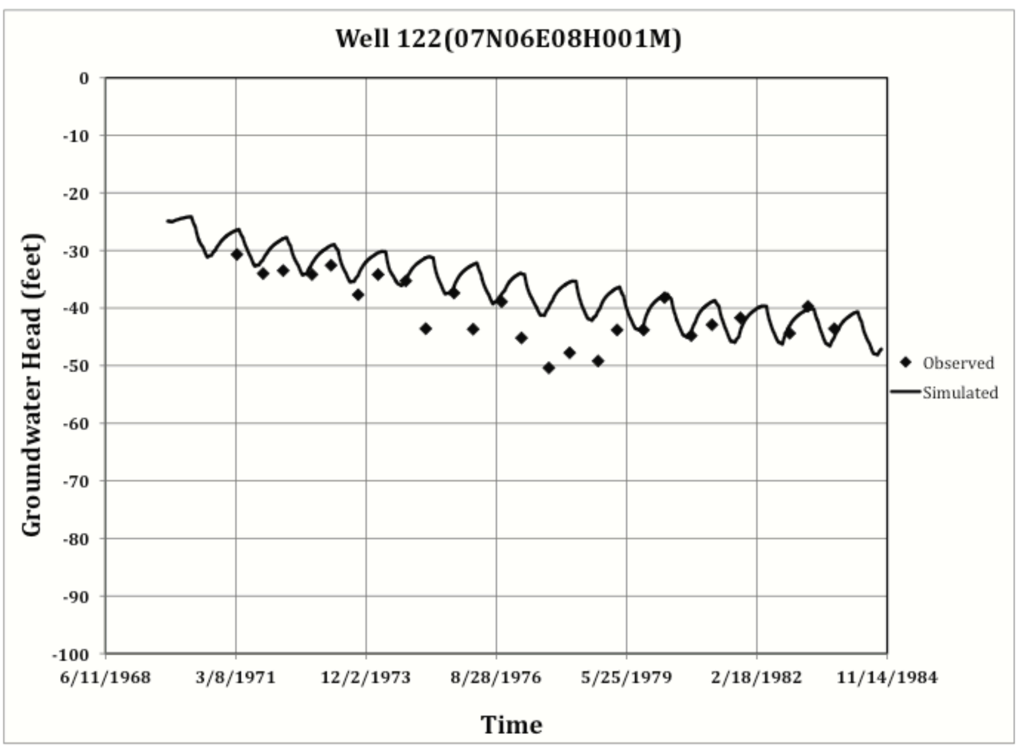

Well 106(07N05E10M001M)
0
-10
-20
-30
-40
-50
-60
-70
-80
-90
-100
Groundwater Head (feet)
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated


Well 112(07N05E26C001M)
Groundwater Head (feet)
0
-10
-20
-30
-40
-50
-60
-70
-80
-90
-100
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated

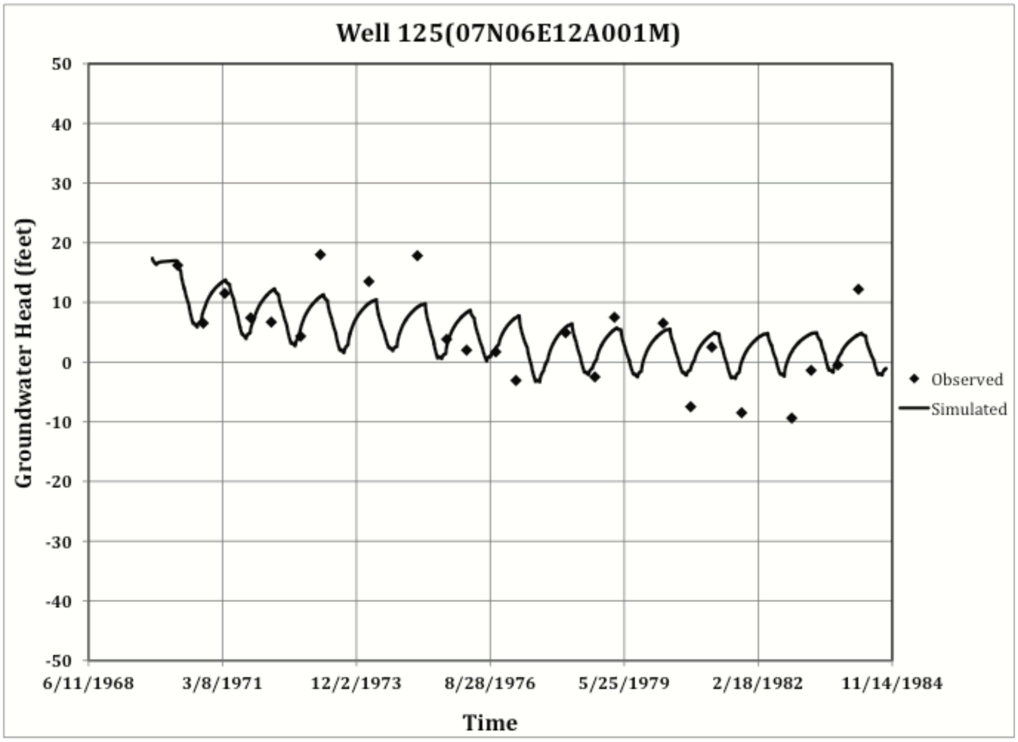

Well 119(07N05E32K001M)
Groundwater Head (feet)
-20
-30
-40
-50
-60
-70
-80
-90
-100
-110
-120
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated

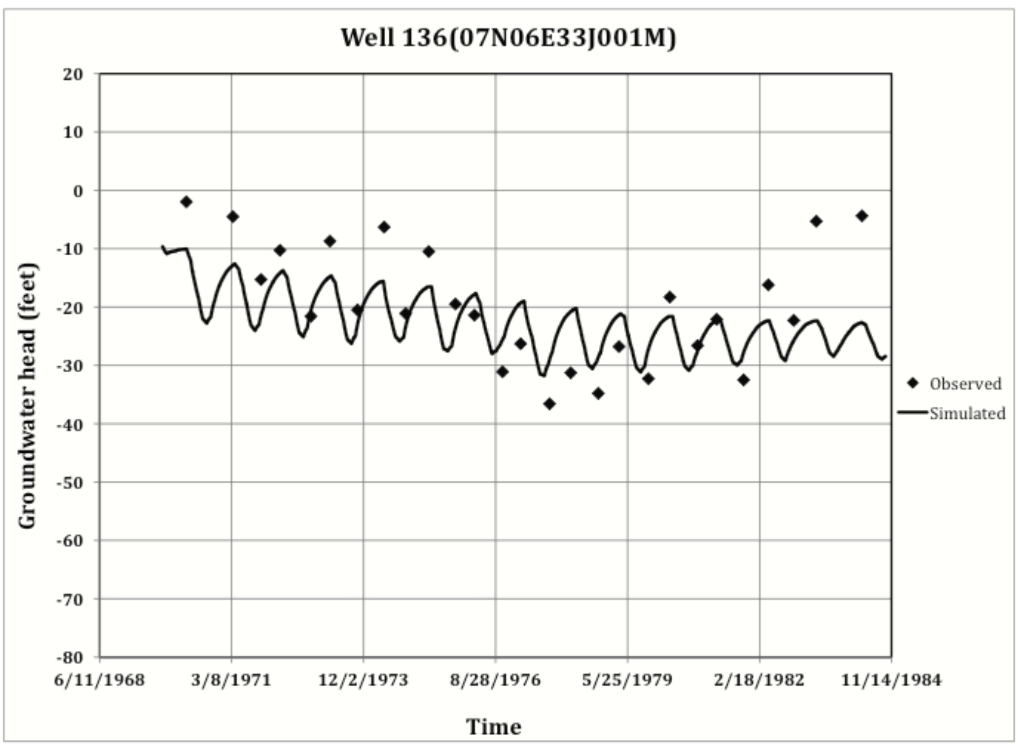


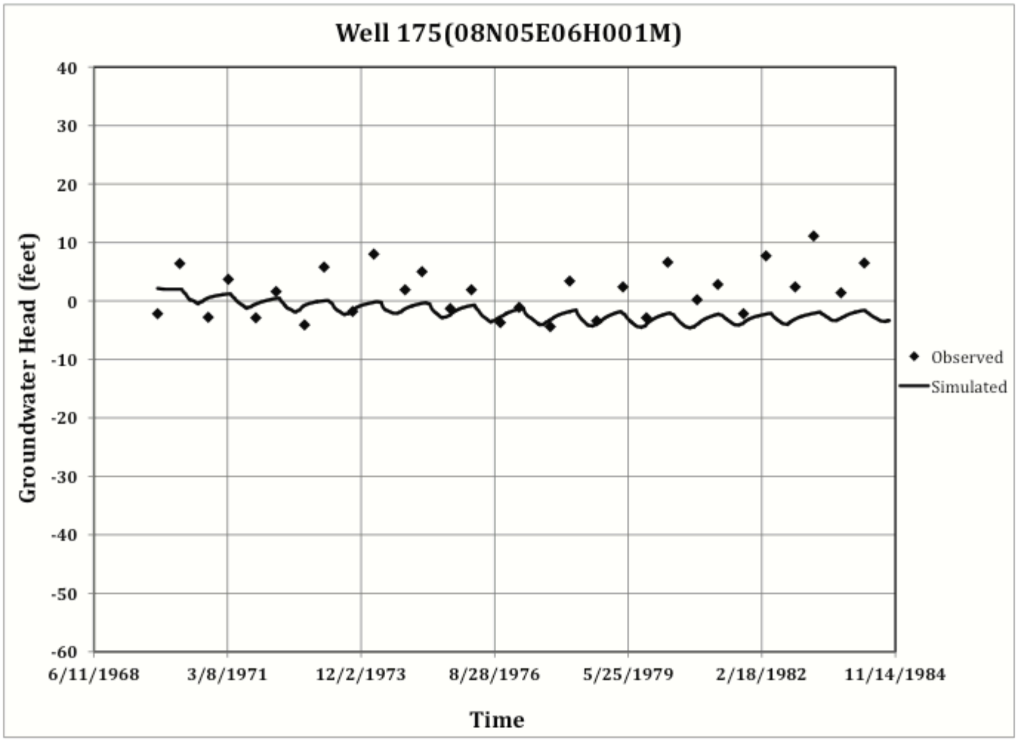
Well 120(07N05E36A001M)
Groundwater head (feet)
-20
-30
-40
-50
-60
-70
-80
-90
-100
-110
-120
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated



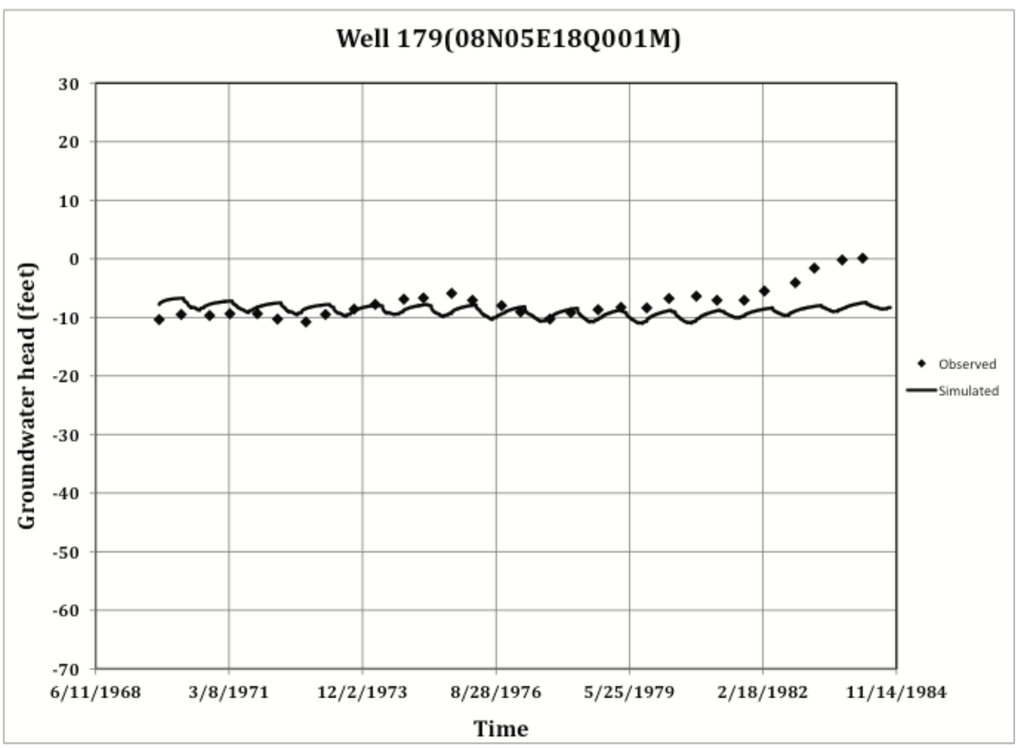
Well 122(07N06E08H001M)
Groundwater Head (feet)
0
-10
-20
-30
-40
-50
-60
-70
-80
-90
-100
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated


Well 125(07N06E12A001M)
Groundwater Head (feet)
50
40
30
20
10
0
-10
-20
-30
-40
-50
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated

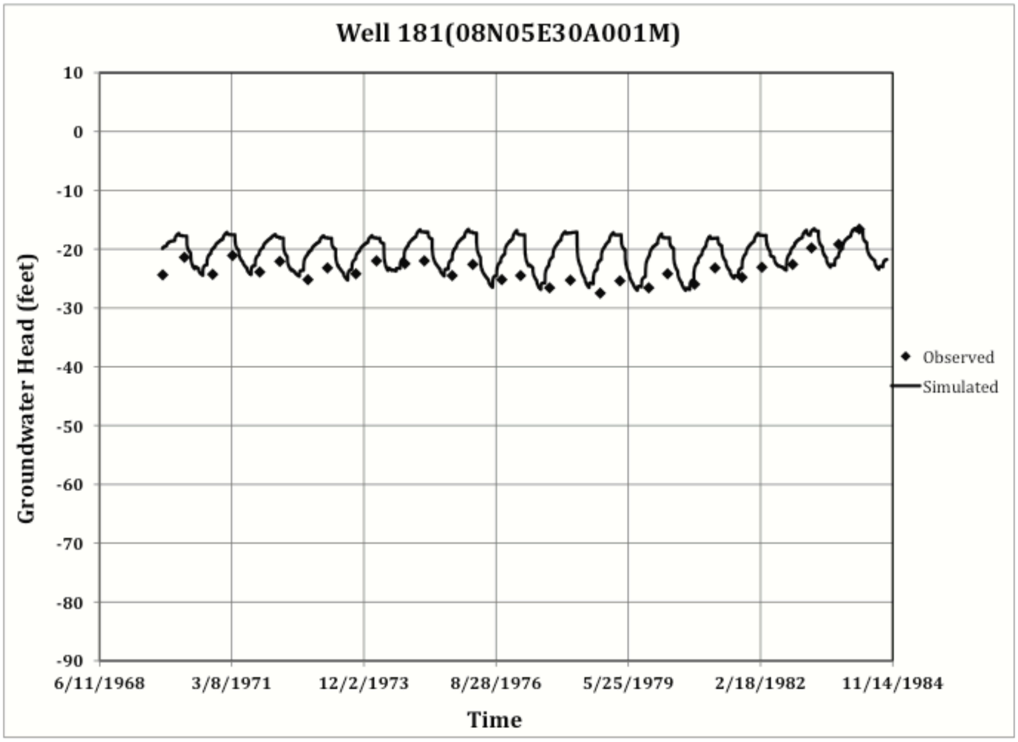

Well 136(07N06E33J001M)
Groundwater head (feet)
20
10
0
-10
-20
-30
-40
-50
-60
-70
-80
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated

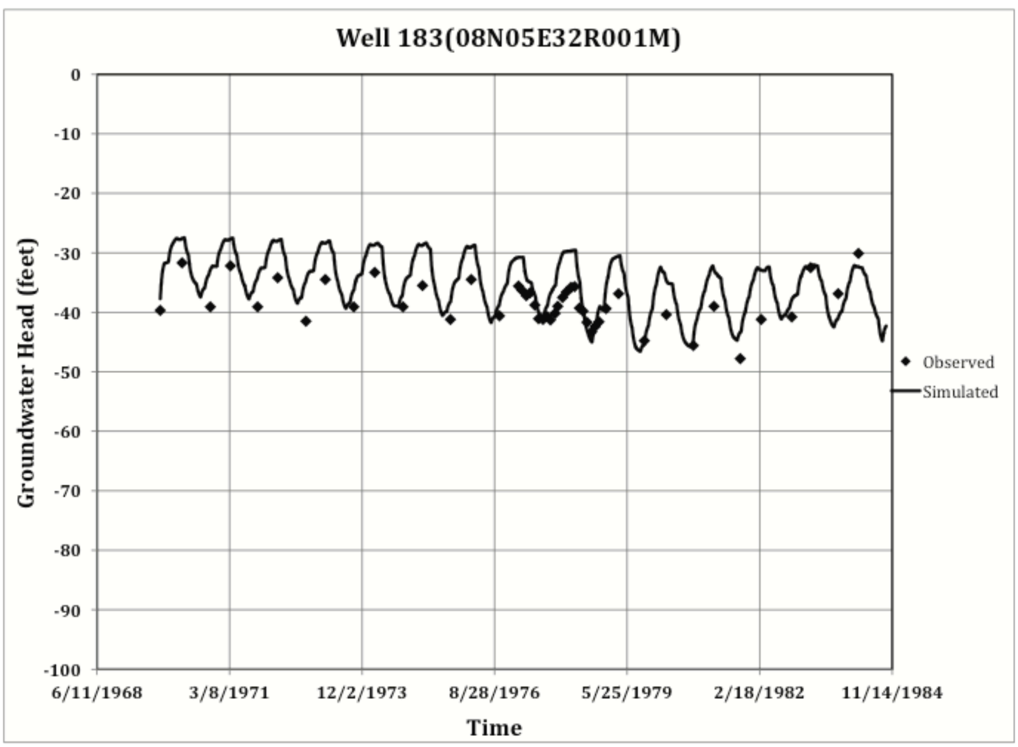


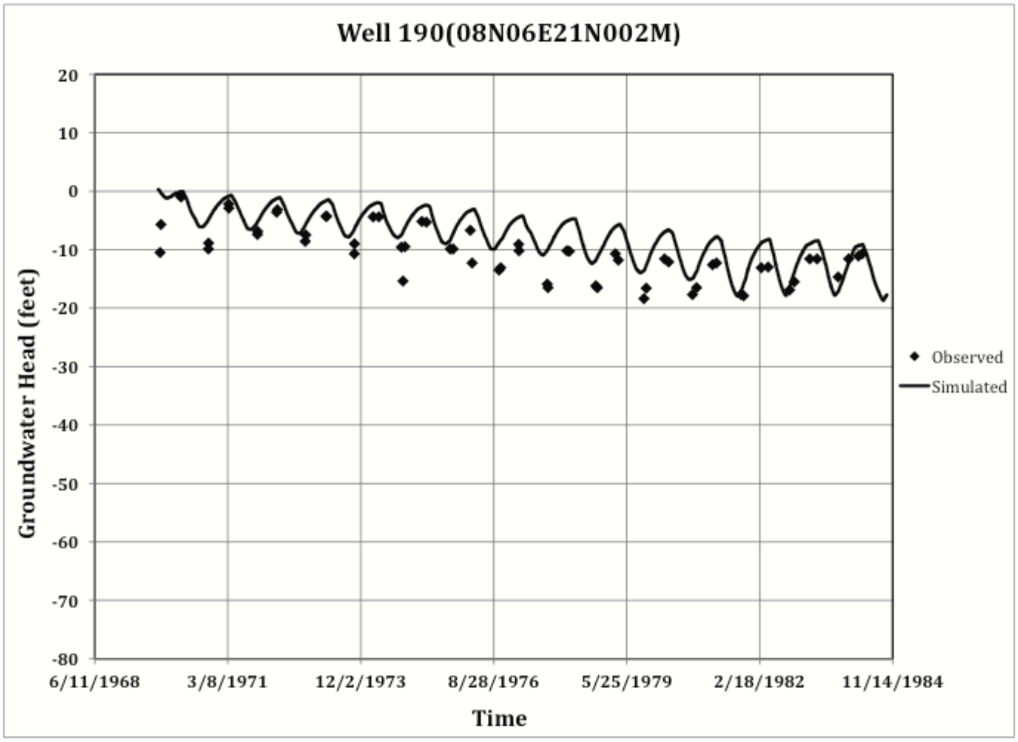
Well 175(08N05E06H001M)
Groundwater Head (feet)
40
30
20
10
0
-10
-20
-30
-40
-50
-60
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated

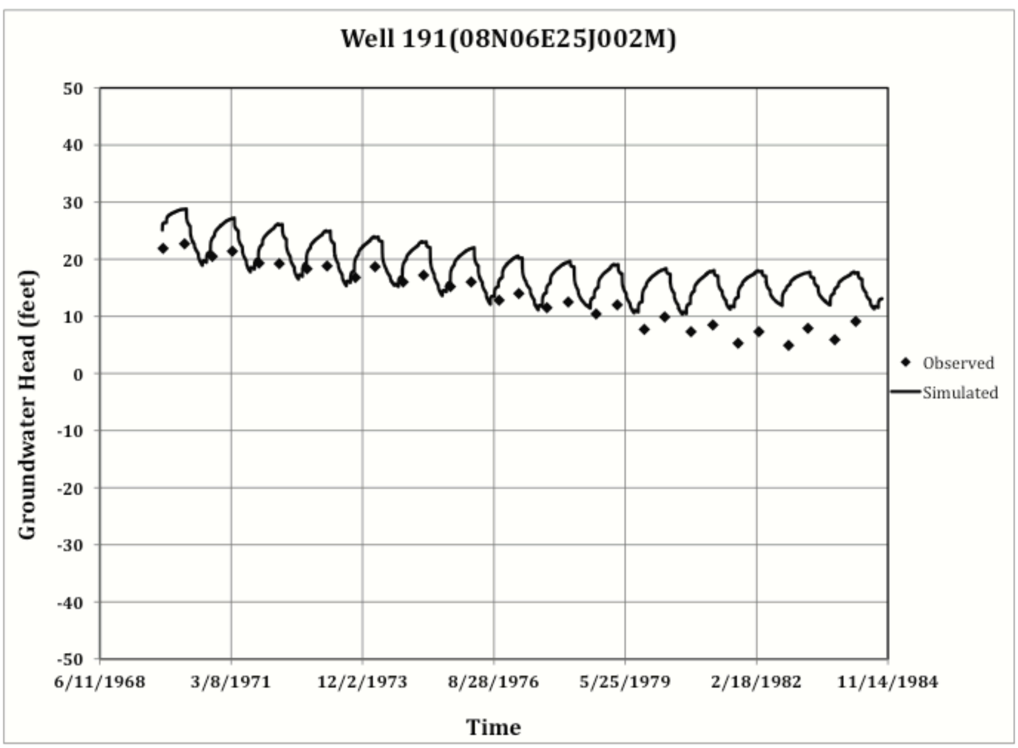


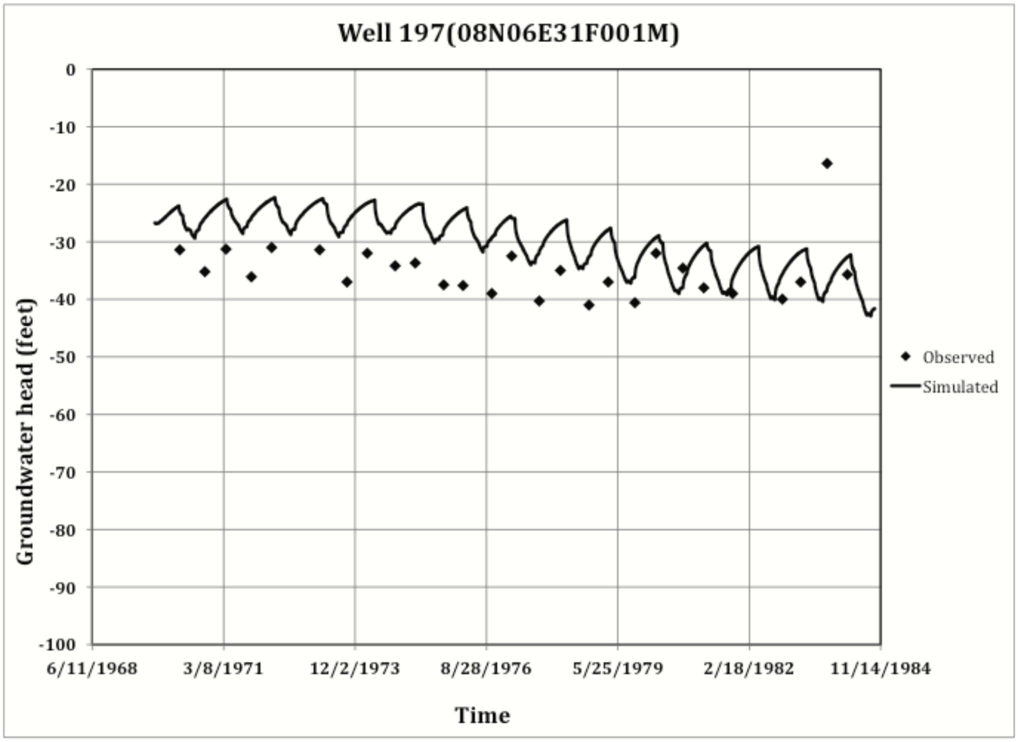
Well 179(08N05E18Q001M)
Groundwater head (feet)
30
20
10
0
-10
-20
-30
-40
-50
-60
-70
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated


Well 181(08N05E30A001M)
10
0
-10
-20
-30
-40
-50
-60
-70
-80
-90
Groundwater Head (feet)
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated


Well 183(08N05E32R001M)
0
-10
-20
-30
-40
-50
-60
-70
-80
-90
-100
Groundwater Head (feet)
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated


Well 190(08N06E21N002M)
Groundwater Head (feet)
20
10
0
-10
-20
-30
-40
-50
-60
-70
-80
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated


Well 191(08N06E25J002M)
Groundwater Head (feet)
50
40
30
20
10
0
-10
-20
-30
-40
-50
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated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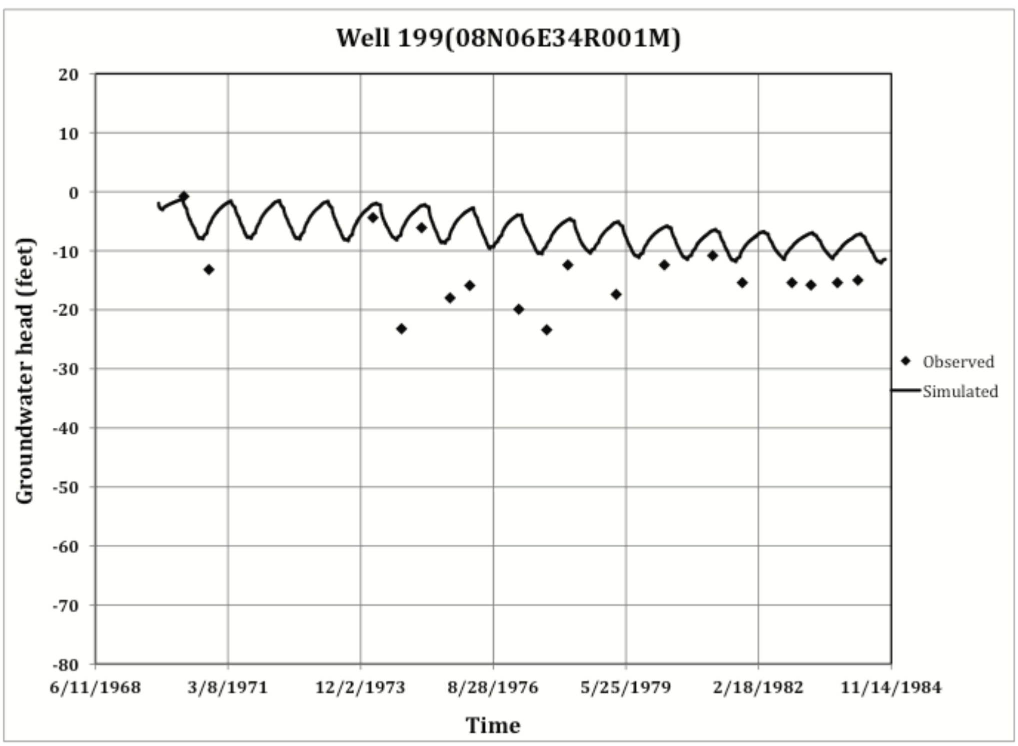
0
-10
-20
-30
-40
-50
-60
-70
-80
-90
-100
Groundwater head (feet)
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated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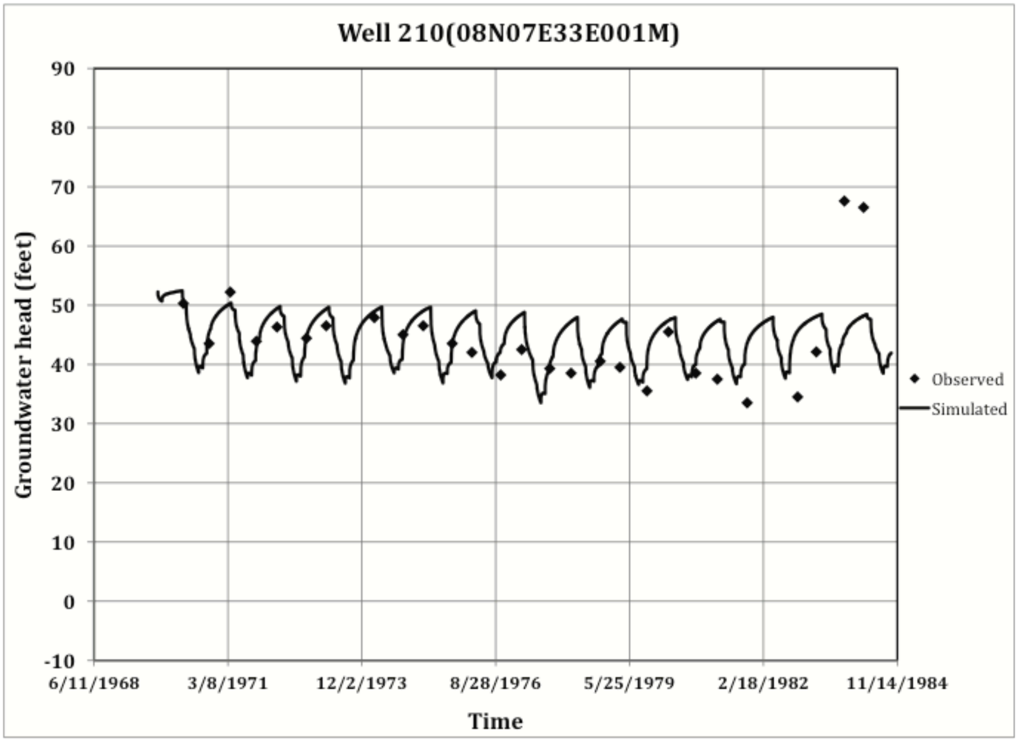
20
10
0
-10
-20
-30
-40
-50
-60
-70
-80
Groundwater head (feet)
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated


Well 210(08N07E33E001M)
Groundwater head (feet)
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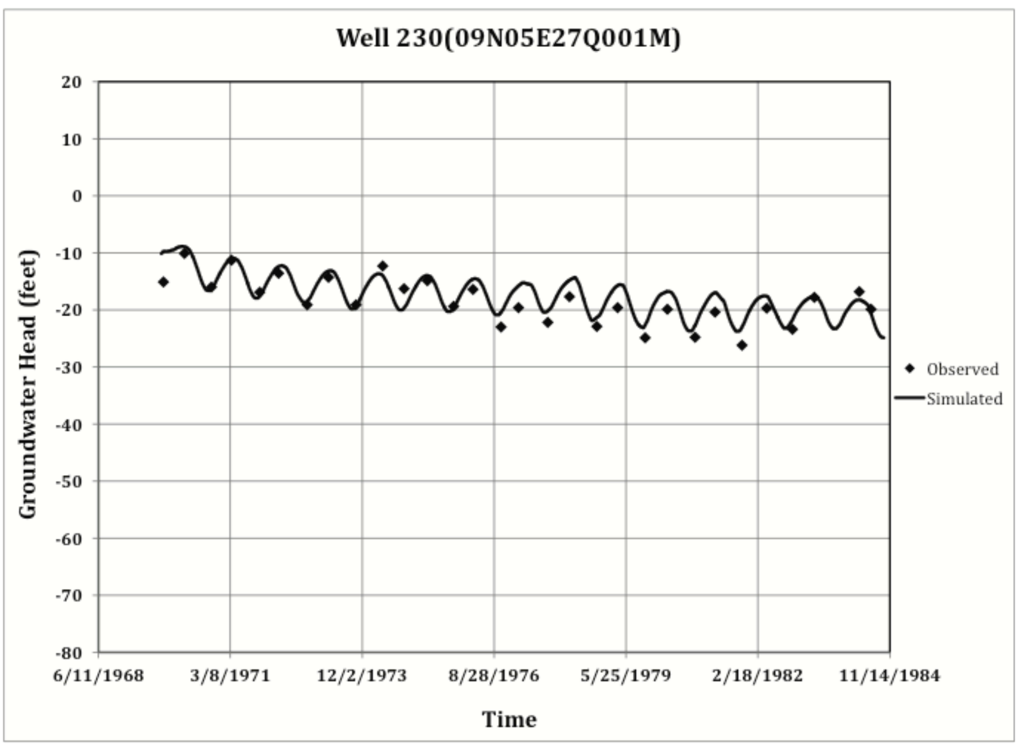
80
70
60
50
40
30
20
10
0
-10
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated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Groundwater Head (feet)
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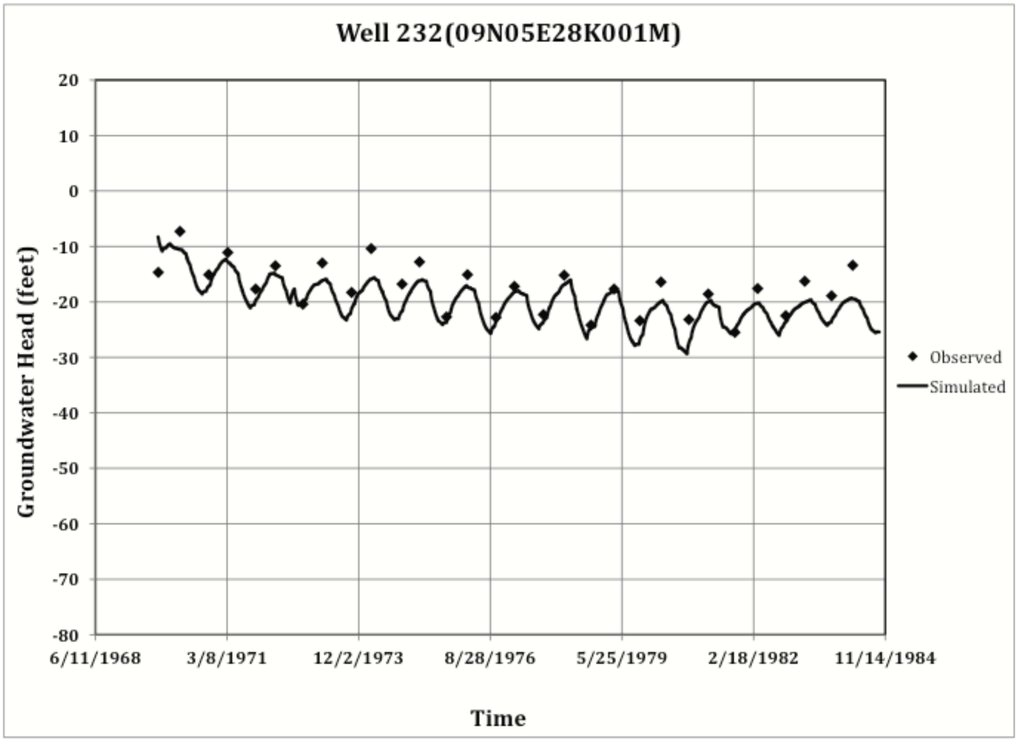
10
0
-10
-20
-30
-40
-50
-60
-70
-80
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated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Groundwater Head (feet)
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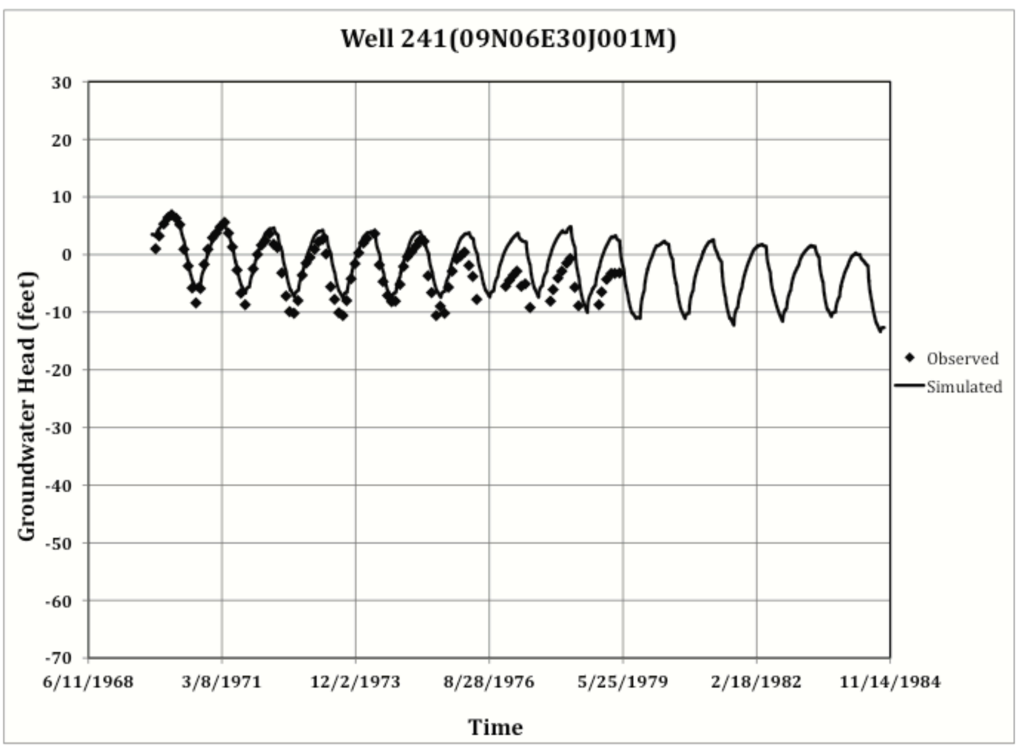
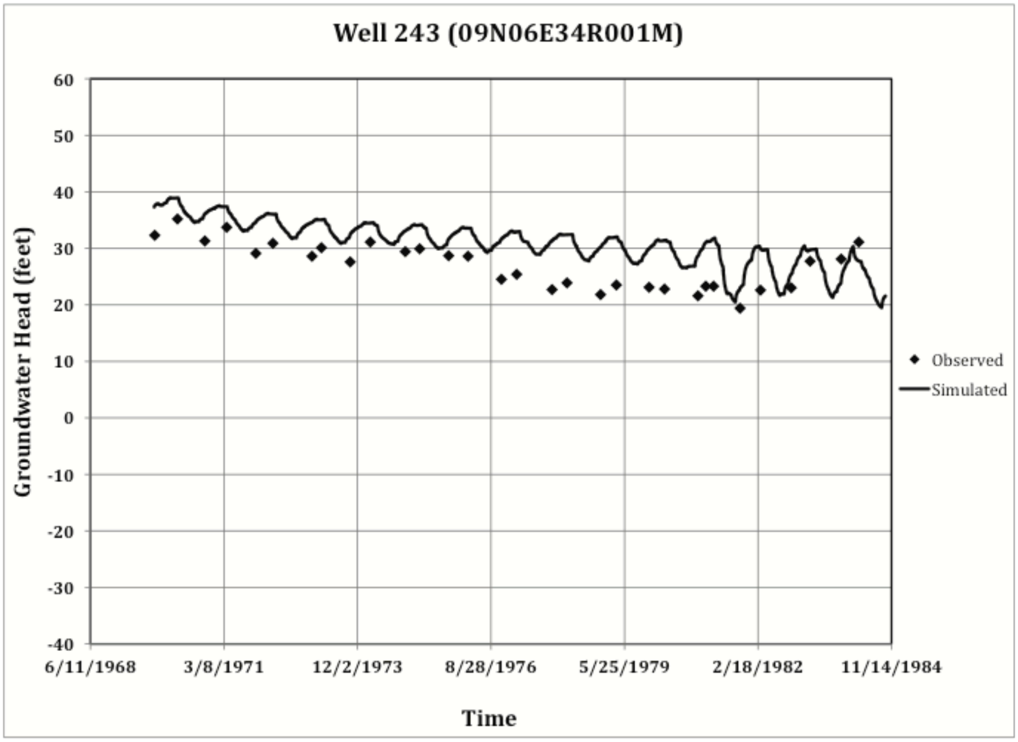
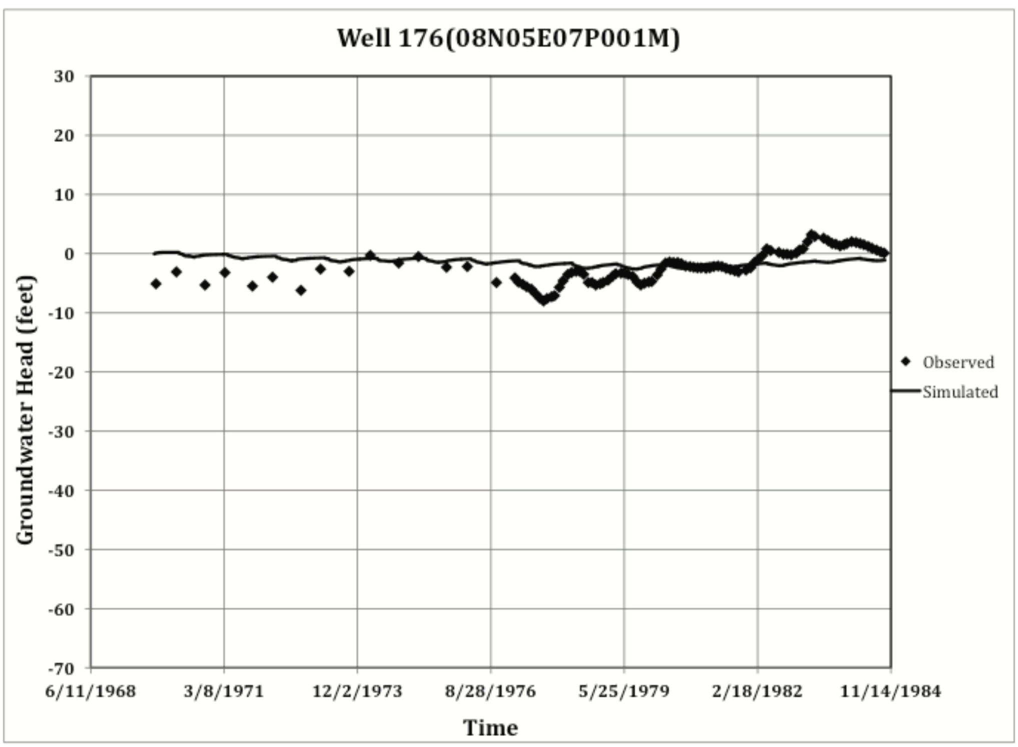
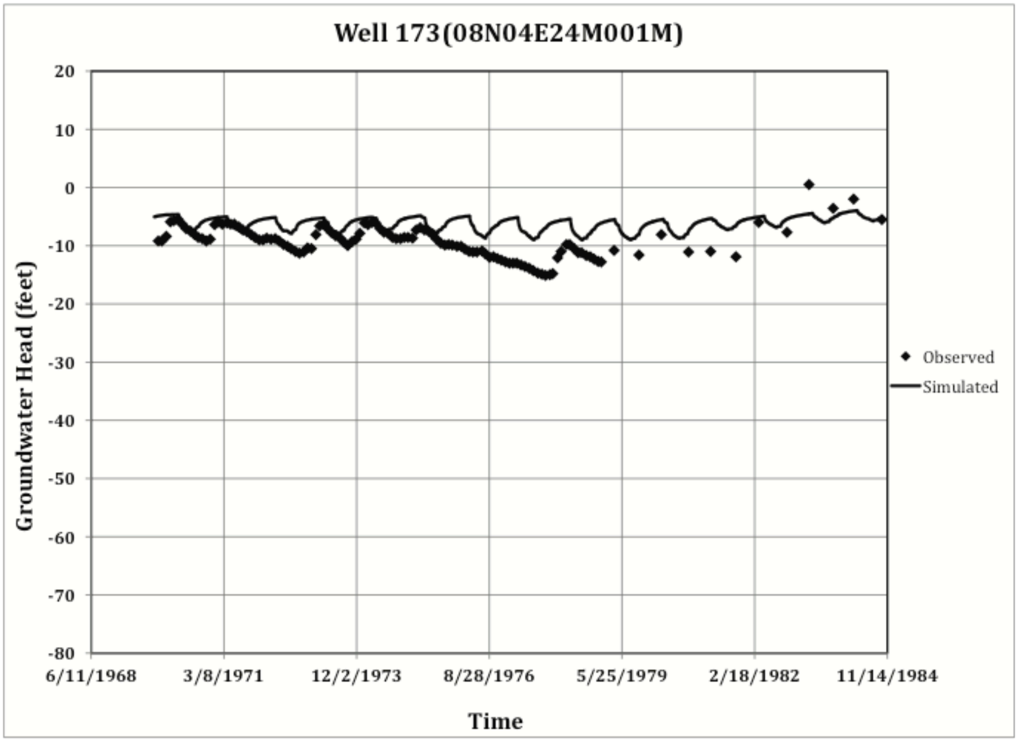
10
0
-10
-20
-30
-40
-50
-60
-70
-80
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated


Well 241(09N06E30J001M)
Groundwater Head (feet)
30
20
10
0
-10
-20
-30
-40
-50
-60
-70
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated


Well 243 (09N06E34R001M)
60
50
40
30
20
10
0
-10
-20
-30
-40
Groundwater Head (feet)
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated


Well 176(08N05E07P001M)
30
20
10
0
-10
-20
-30
-40
-50
-60
-70
Groundwater Head (feet)
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated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Groundwater Head (feet)
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10
0
-10
-20
-30
-40
-50
-60
-70
-80
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated

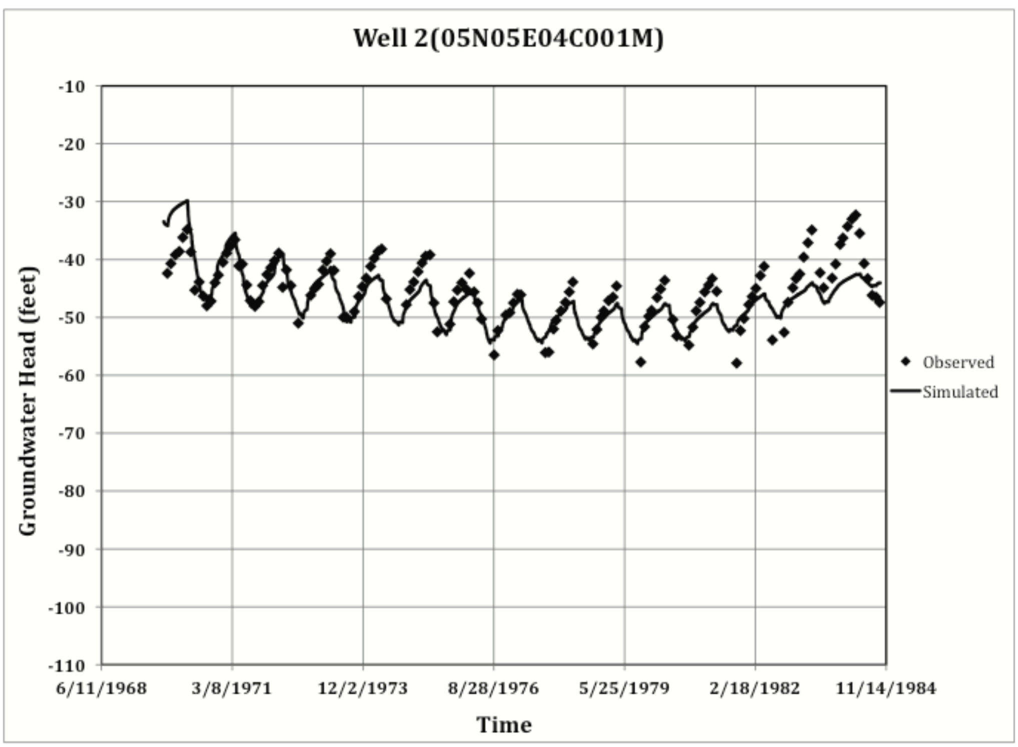
Well 2(05N05E04C001M)
Groundwater Head (feet)
-10
-20
-30
-40
-50
-60
-70
-80
-90
-100
-110
6/11/1968
3/8/1971
12/2/1973
8/28/1976
5/25/1979
2/18/1982
11/14/1984
Time
Observed
Simulated

Figure D.4 comparison of simulated and observed groundwater head at selected calibration wells

Conservation Biology

Special Section

Combined Effects of Climate Change and Bank Stabilization on Shallow Water Habitats of Chinook Salmon

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Abstract: *Significant challenges remain in the ability to estimate habitat change under the combined effects of natural variability, climate change, and human activity. We examined anticipated effects on shallow water over low-sloped beaches to these combined effects in the lower Willamette River, Oregon, an area highly altered by development. A proposal to stabilize some shoreline with large rocks (riprap) would alter shallow water areas, an important habitat for threatened Chinook salmon (Oncorhynchus tshawytscha), and would be subject to U.S. Endangered Species Act-mandated oversight. In the mainstem, subyearling Chinook salmon appear to preferentially occupy these areas, which fluctuate with river stages. We estimated effects with a geospatial model and projections of future river flows. Recent (1999–2009) median river stages during peak subyearling occupancy (April–June) maximized beach shallow water area in the lower mainstem. Upstream shallow water area was maximized at lower river stages than have occurred recently. Higher river stages in April–June, resulting from increased flows predicted for the 2080s, decreased beach shallow water area 17–32%. On the basis of projected 2080s flows, more than 15% of beach shallow water area was displaced by the riprap. Beach shallow water area lost to riprap represented up to 1.6% of the total from the mouth to 12.9 km upstream. Reductions in shallow water area could restrict salmon feeding, resting, and refuge from predators and potentially reduce opportunities for the expression of the full range of life-history strategies. Although climate change analyses provided useful information, detailed analyses are prohibitive at the project scale for the multitude of small projects reviewed annually. The benefits of our approach to resource managers include a wider geographic context for reviewing similar small projects in concert with climate change, an approach to analyze cumulative effects of similar actions, and estimation of the actions' long-term effects.*

Keywords: chinook salmon, endangered species act, mainstem, riprap, riverbank stabilization, section 7 consultation, Willamette river

Efectos Combinados del Cambio Climático y la Estabilización de Bordes de Ríos Hábitats de Aguas Poco Profundas del Salmón Chinook

Resumen: *Todavía permanecen obstáculos significativos en la habilidad para estimar el cambio de hábitat bajo los efectos combinados de la variabilidad natural, el cambio climático y la actividad humana. Examinamos los efectos anticipados en el agua poco profunda sobre playones con poca inclinación a estos efectos combinados en la parte baja del río Willamette, Oregon, un área altamente alterada por el desarrollo. Una propuesta para estabilizar algunos bordes con rocas grandes (escolleras) alteraría las áreas de poca profundidad, un hábitat importante para el salmón Chinook (Oncorhynchus tshawytscha), una especie amenazada, y estaría sujeta a revisiones mandadas por el Acta Estadunidense de Especies Amenazadas. En el cauce principal, salmones menores al año parecer ocupar preferencialmente áreas que fluctúan con etapas de río. Estimamos los efectos con un modelo geoespacial y proyecciones futuras de caudales de río.*

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La media de las etapas de río recientes (1999–2009) durante ocupaciones críticas de salmones menores al año (abril-junio) maximizó el área de playones con poca profundidad en la parte baja del cauce principal. El área de poca profundidad río arriba se maximizó más en etapas más bajas del río de lo que ha ocurrido recientemente. Etapas más altas del río en abril-junio, resultantes de incrementos de flujo predichos para los 2080s, disminuyeron el área de playones de poca profundidad de 17–32%. Con base en los flujos proyectados para 2080, más del 15% del área de playones de poca profundidad fue desplazada por la escollera. El área de playones de poca profundidad perdida por la escollera representó hasta el 1.6% del total de la boca del río hasta 12.9 Km río arriba. Las reducciones en el área de playones de poca profundidad pueden restringir la alimentación de los salmones, sus descansos y refugios contra depredadores y reducir potencialmente las oportunidades de expresión del rango total de estrategias de historias de vida. Aunque el análisis del cambio climático proporcionó información útil, los análisis detallados son prohibitivos en la escala de proyecto para la multitud de proyectos pequeños revisados anualmente. Los beneficios de nuestro estudio para los administradores de recursos incluyen un contexto geográfico más amplio para revisar proyectos pequeños similares en relación con el cambio climático, una aproximación para analizarlos efectos acumulativos de acciones similares y la estimación de los efectos a largo plazo de las acciones.

Palabras Clave: Acta de Especies Amenazadas (ESA), consultoría Sección 7, escollera, estabilización de orillas de ríos, río Willamette, salmón Chinook, tallo principal

Introduction

Understanding the effects of climate change presents a significant challenge to natural resource managers. A compounding factor to this challenge is that species often use different habitats at different life stages. For example, anadromous Pacific salmon can use different habitat for spawning, rearing, and migration. Changes in salmon freshwater habitat can occur due to deterministic and stochastic events at a variety of temporal and spatial scales (Minns et al. 1996; Montgomery & Buffington 1998). The structure and function of fish habitat reflects large-scale basinwide characteristics and habitat-forming processes at the basin and reach scales (Naiman & Bilby 1998). Habitat alterations can have dramatic effects on species diversity. Given the dynamic nature of habitat-forming processes and variability in fish responses to climate change (e.g., Crozier & Zabel 2006), it is difficult to predict species persistence.

Anadromous salmonids, because of their high degree of philopatry, exhibit strong local adaptation to the habitats they occupy (Quinn 2005); thus, habitat changes can have substantial effects on survival (e.g., Mantua et al. 2010). Advances in climate modeling allow investigators to predict changes to freshwater systems. Changes in the timing and magnitude of stream flows, water temperatures, and other factors such as vegetation, will affect the quantity, quality, composition, and complexity of freshwater habitat (e.g., Adams et al. 2009; Chang & Jones 2010; Mote & Salaté 2010).

Coincident with large-scale, long-term climate forces that reshape habitat, is a myriad of smaller scale anthropogenic habitat alternation activities. In the United States, the Endangered Species Act (ESA) mandates that any government projects or actions that have the potential to affect habitat of ESA-listed species must be reviewed by either the National Marine Fisheries Service (NMFS) or U.S. Fish and Wildlife Service (USFWS) prior to project implementation (Seney et al. 2013). Section 7(a)(2) (hereafter Section 7) of the ESA states that each federal agency shall insure that any action they authorize, fund, or carry out is not likely to jeopardize the continued existence of a listed species or result in destruction or adverse modification of designated critical habitat. During a review, NMFS or USFWS responds with either a concurrence letter for projects determined to have insignificant effects or a biological opinion for projects likely to cause adverse effects. The agencies must use the best available scientific information when reviewing proposed projects. This includes information, when relevant, about predicted climate change and its corresponding effects on the species' habitats. Currently, some biological opinions include general information on climate change forecasts at the regional and watershed-basin scale. However, opinions usually do not translate large-scale forecasts down to the scale of the project or use climate information to analyze the action's effects into the future. These mismatches in scale (e.g., Lewis et al. 1996) are largely due to a lack of downscaled information in a form useful to evaluate project effects at multiple spatial and temporal scales.

We examined changes in habitat of endangered Chinook salmon (*Oncorhynchus tshawytscha*) under climate change and a proposed streambank stabilization project within the context of an ESA Section 7 project review. We predicted there would be a change in shallow-water area along low-sloped beaches as a function of river stage levels. We incorporated river stages estimated for forecasts of river flow for the 2080s. We sought to determine how the amount and temporal extent of shallow-water habitat changed with river level; the effect of a small-scale action on habitat; whether the inclusion of climate change considerations changed the action's effect at the project scale (within the project's footprint) and river reach scale beyond the bounds of the project's footprint; and implications for how the action, in concert with climate change, might affect species viability.

Lower Mainstem Willamette River Stream-Bank Stabilization

Although a few Section 7 consultations are large and complex (e.g., operation of the federal dams on the Columbia River), the vast majority of projects subject to consultation are small, such as dock construction or placement of large rocks (riprap) to stabilize a section of stream bank. We focused on a proposed project to place riprap along approximately 450 m of shoreline in the Portland Harbor area near river km (Rkm) 3.2 (Fig. 1). Riprap is a common bank armoring method used to stabilize riverbanks and to prevent channel migration and thus to reduce bank erosion. However, it impairs ecological processes, disrupts surface and subsurface flow exchange, and inhibits development of streamside vegetation (Fischenich 2003). Alteration of these processes affects gravel recruitment and development of habitat for fishes, and the cumulative effects of additional armoring is largely unexplored (Schmetterling et al. 2001).

The Willamette River is the tenth largest river in the contiguous United States as measured by discharge, and it flows approximately 500 km northward from its origin in the Cascade Range to its confluence with the Columbia River (Kammerer 1990). Riverbanks were historically dominated by extensive mixed deciduous and coniferous gallery forests (Gregory et al. 2002). The river has changed dramatically as a result of development activities in the basin. Since the mid-1800s the bathymetry of the lower river's channel has been modified to accommodate development and large vessel shipping traffic. Numerous heavy industries, chemical plants, and port facilities have operated along the shore of the river, resulting in the addition of Portland Harbor (Rkm 3.2–19.0) to the Superfund National Priorities list in 2000. Bank armoring (e.g., riprap, bulkheads, sheet piles) comprises about 11% of the shoreline below Rkm 64 (Hughes & Gammon 1987).

The Willamette River basin is home to 5 ESA-listed anadromous salmonid evolutionarily significant units (ESUs). We focused on Upper Willamette River (UWR) Chinook salmon (NMFS 2005), which inhabit the mainstem Willamette River nearly all months of the year as upstream migrating adults and downstream migrating and rearing juveniles (Mattson 1962; Friesen et al. 2007). They primarily exhibit a yearling (stream-type) juvenile life history, with juveniles spending about 1 year rearing in upstream tributaries before migrating to the estuary and ocean (Myers et al. 2006). Yearlings typically migrate through the Portland Harbor reach in 4–6 d, feeding and growing during their passage (Friesen et al. 2007). However, a substantial number of juveniles leave their natal upstream areas and migrate to the mainstem as subyearlings (Mattson 1962). Little is known about the duration of subyearling residence or their preferred habitat in the mainstem (Friesen et al. 2007). In the lower mainstem, as in other areas of the Columbia River basin (Dauble et al. 1989; Garland et al. 2002) and elsewhere (Tabor et al. 2011), subyearling emigrants appear to occupy primarily shallow water along low-sloped beaches (hereafter SWH) (Ward et al. 1994). In recent surveys of the lower mainstem, subyearlings were caught almost exclusively in beach seines at SWH sampling sites rather than through electrofishing in midchannel or other areas that were not SWH (Friesen et al. 2007). Although gear bias could be a contributing factor for their appearance only in the beach seines, mainstem SWH appear to be important for subyearlings (Friesen et al. 2007).

Methods

Study Area and Geospatial Model

Using an approach similar to that of Kukulka and Jay (2003), we evaluated habitat changes in the area below Willamette Falls downstream to the confluence with the Columbia River (Fig. 1) in response to surface water elevations in the river, or river stages. We defined SWH as the area, for shorelines characterized as beach (low slope with small sand and silt substrate; Supporting Information), from the shoreline out to a water depth of 3 m. This SWH depth range is consistent with occurrence of mainstem juvenile Chinook salmon (Friesen et al. 2007) and results of investigations of subyearling habitat use (Dauble et al. 1989; Tiffan et al. 2002; Tabor et al. 2011). We subdivided the study area into 4 sections by visual inspection at transitions in relative river sinuosity and pool width: section 1, Rkm 0–12.9; section 2, Rkm 12.9–25.8; section 3, Rkm 25.8–33.8; and section 4, Rkm 33.8–42.6 (Fig. 1).

We developed a geospatial representation of the lower Willamette River and channel system to estimate river stage and riprap effects on SWH. We combined topographic and bathymetric data into a 1-m grid cell resolution digital terrain model (DTM) of the lower Willamette River and floodplain which was developed from recent and historical National Oceanographic and Atmospheric Administration hydrographic surveys of riverbed bathymetry and a lidar-derived (light detection and ranging) bare earth digital elevation model of shoreline and upland surface topography (Supporting Information). We converted the DTM from North American Vertical Datum of 1988 to the Columbia River Datum (CRD) with a geoid separation model (Supporting Information). All river stage elevations in this study are presented relative to CRD.

We estimated the quantity of SWH at different river levels in the DTM. We filled the DTM with water by

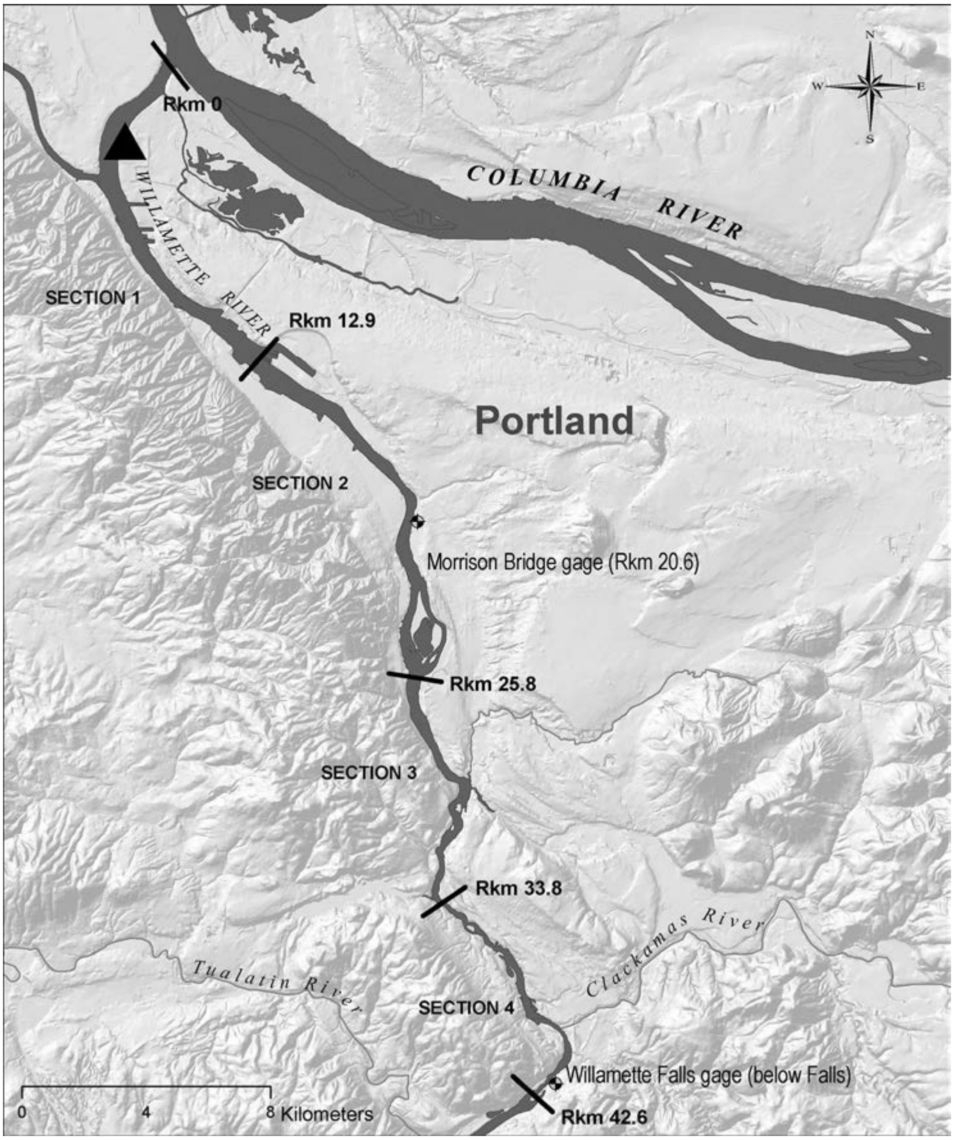

Figure 1. The lower mainstem of the Willamette River. River stage gages were located at the Morrison Street Bridge and just below Willamette Falls. Sections 1–4 are referred to in the study, and the approximate location of a proposed riprap project is marked by the triangle.

creating water surface elevation cross-sections spaced 0.8 km apart and linearly interpolated between the cross-sections to create water surface grids. We constructed a range of water surface elevations (i.e., stages) in 0.6-m increments that spanned historical and recently observed April–June river stages and those anticipated as a consequence of flows in the future. Observations came from stage data measured at Morrison Bridge (Rkm 20.6, U.S. Geological Survey [USGS] flow gauge 14211720), below Willamette Falls (Rkm 42.2, USGS flow gage 14207770), ordinary high water elevations (USACE 2004), and gradient-based water surface changes (USACE 2004) (Supporting Information). Each water surface elevation grid (i.e., stage) was subtracted from the DTM of the river bed, and we converted the results to SWH polygons, each capturing the area within the 0–3-m depth range.

To estimate the effects of a riverbank stabilization project on SWH, we superimposed a hypothetical riprap installation onto the DTM near the proposed project area (Fig. 1). We estimated the installation's footprint with a 150 × 470 m polygon that covered an area adjacent to and within the wetted channel of the river. We estimated SWH displacement by the footprint for each river stage.

We focused on the range of river stages in the period during the April–June peak of subyearling occupancy (Ward et al. 1994; Friesen et al. 2007). We estimated SWH for 3 timeframes: recent (1999–2009), historical (before construction of large mainstem dams that affected flows and stages [i.e., before 1937]), and the 2080s. The recent stages consisted of monthly medians of the mean daily river stages (Fig. 2). Historical stages were monthly medians from once-per-day observations at the Morrison Bridge location (USACE 2004) (Supporting Information).

The 2080s stages were predicted from a simple model of river stage as a function of future flow predictions in the Columbia and Willamette Rivers (Hamlet et al. 2010) (Fig. 2). The model, $\text{stage}_{(\text{Morrison Bridge})} = -0.286 +$

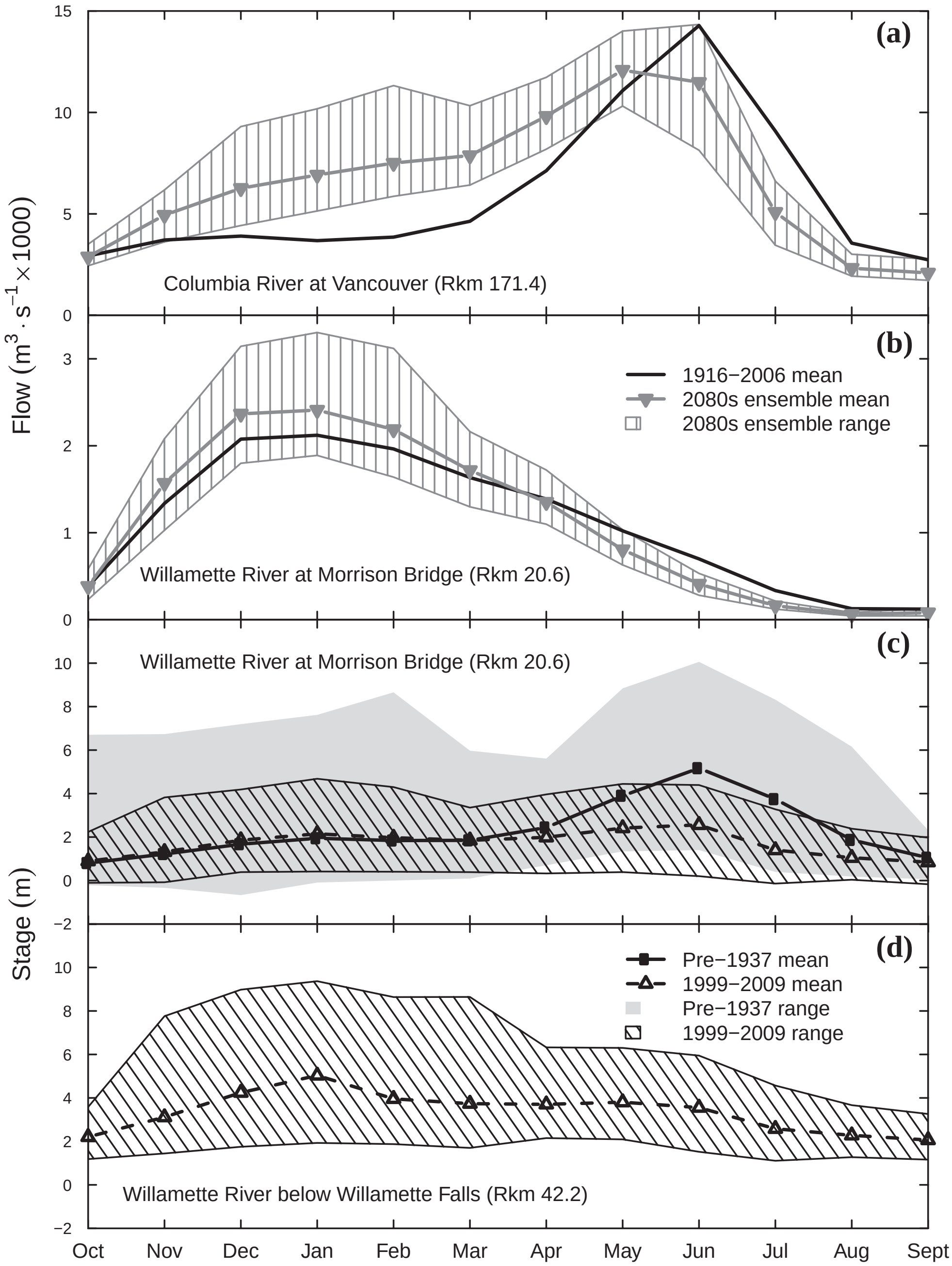

Figure 2. Monthly flows for the (a) Columbia and (b) Willamette Rivers during the contemporary era (1916–2006), and the mean and range of flows predicted for the 2080s from an ensemble of hydrology model outputs that used results from 10 global climate models under the A1B climate scenario (Hamlet et al. 2010). River stages of the Willamette River have been recorded at the (c) Morrison Bridge (Rkm 20.6; historical and recent) and (d) below Willamette Falls (Rkm 42.2; recent period only).

$0.507 \times \text{flow}_{\text{(Willamette)}} + 0.292 \times \text{flow}_{\text{(Columbia)}}$ ($R^2 = 0.93$, $p < 0.0001$), was derived from recent April–June flow records for the Willamette River at the Morrison Bridge gage and from a gage on the Columbia River at Vancouver, Washington (Rkm 171.4, USGS flow guage 14144700). For simplicity, the model did not incorporate tidal influence on river stage, which is generally weak during the spring flow period (Kukulka & Jay 2003). The 2080s April–June stages were predicted with the April–June monthly means of flows from an ensemble of hydrology model outputs that estimated flows for 10 global climate models (GCMs) under the A1B greenhouse gas (GHG) emissions scenario (Hamlet et al. 2010). Given the uncertainty of future climate, an ensemble of 10 GCMs captures the consensus of the trends, and the mean of an ensemble generally outperforms any of an ensemble's constituent

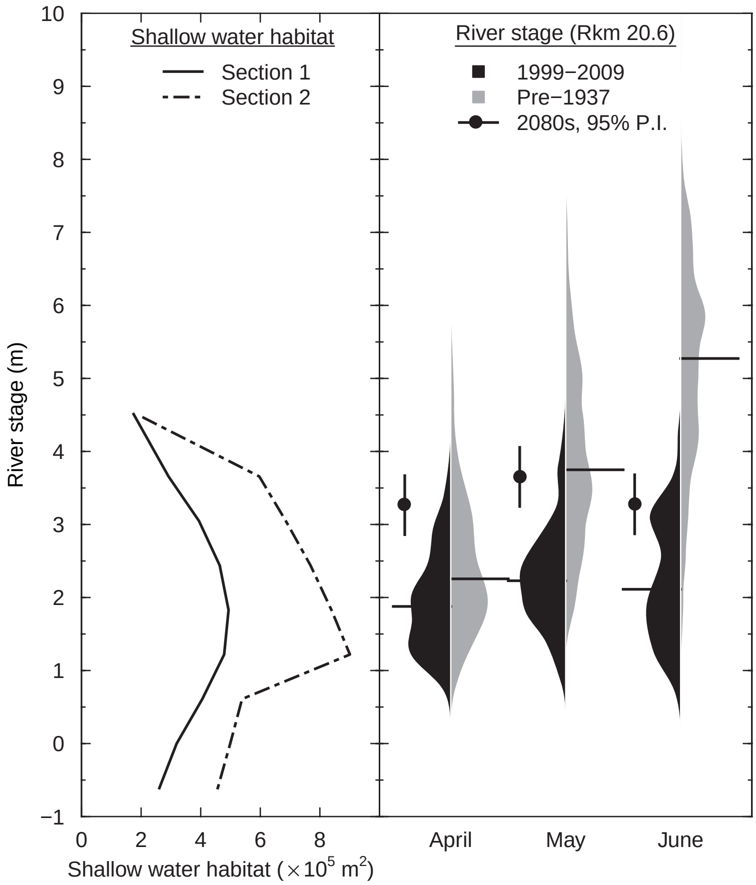

Figure 3. Area of shallow water along low-sloped beaches (SWH) (left panel) for Willamette River mainstem sections 1 and 2 (Fig. 1) as a function of river stage. Distributions of April–June river stages (right panel) at the Morrison Bridge during recent (1999–2009) and historical (pre-1937) periods (median stages are indicated by the horizontal bars). Predictions of 2080s river stages at Morrison Bridge (right panel, with 95% prediction intervals) came from a regression model of river stage as a function of Willamette and Columbia Rivers' flows, where flows were the ensemble average from predicted flows from a hydrology model run with inputs from 10 global climate models (Hamlet et al. 2010).

members (Mote et al. 2011; Snover et al. 2013). Among the 2 GHG emissions scenarios that Hamlet et al. (2010) evaluated in their study (A1B and B1), we chose their flow estimates from the A1B scenario because it represents a medium emissions scenario, whereas B1 represents significant GHG mitigation by the close of the 21st century (Mote & Salaté 2010). We presented estimates of SWH over a relatively large range of river stages to account for the uncertainty in climate, streamflow projections, and dam operations upstream of the study area, all of which contribute to river stage.

Results

The area of SWH was affected by changes in river stage, particularly between the lower and middle river stages. There was a rapid increase in SWH as river stage increased (Figs. 3 & 4). The area of SWH nearly doubled as river stages moved from the low to middle stages and reached a peak in the middle river stages. In the study reach as a whole, as river stage increased from middle to higher stages there was a gradual decline in SWH, and SWH tailed off sharply at the ordinary high water mark. The lower mainstem (sections 1 and 2) contained the majority of SWH (approximately 75%) and was the main driver of SWH fluctuations (Figs. 3 & 4).

Spatial Extent of SWH

Recent median river stages were at or near levels that maximized SWH in the lower mainstem. Beach SWH was more abundant in the lower 2 sections and reached a maximum in the lower 2 sections at river stages between +1 to +3 m (Fig. 3). Maximum SWH occurred at river stages of nearly +2 m in section 1, the area that included the proposed project site. In section 2, river stages of just over +1 m maximized the amount of SWH. Recent (1999–2009) median river stages during April–June were near +2 m, corresponding to the peak of section 1 SWH and above the peak of SWH in section 2 (Fig. 3). Sections 3 and 4 contained less SWH than the lower sections at most stage levels (Fig. 4) and had broad plateaus of SWH from +1 m up to stages of +3.5 m (Fig. 4). Section 3 SWH was roughly maximized between river stages +2 to +3 m. Section 4 had a broad plateau of SWH between +1.5 and +3.5 m. Median river stage below Willamette Falls during April–June was about +3.5 m, which was in the upper range of the SWH plateau (Fig. 4).

Median river stages were higher before 1937 during April–June relative to the recent period, about 0.5–3.5-m higher than recent stages, and June river stages often exceeded +5 m (Fig. 3) (modern flood stage at Morrison Bridge is +5.5 m). High historical spring river stages resulted in overbank flows and, in combination

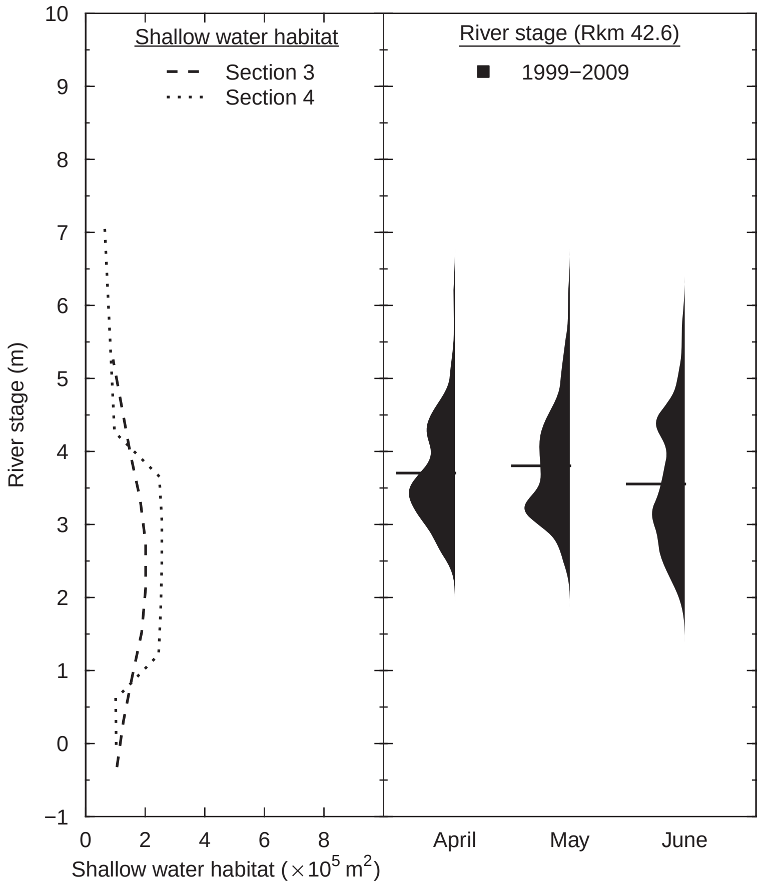

Figure 4. Area of shallow water along low-sloped beaches (SWH) (left panel) for Willamette River mainstem sections 3 and 4 (Fig. 1) as a function of river stage. Distributions of recent (1999–2009) April–June river stages (right panel) measured below Willamette Falls. Median stages are indicated by the horizontal bars.

with connections to the historical floodplain, created opportunities for SWH formation. We are unaware of historical river stage data for the upper 2 river sections comparable to the historical data for Morrison Bridge.

Effects of climate change and the riprap project on SWH extent varied according to river stage. River stages in the 2080s could inundate much of the SWH throughout the lower Willamette River. For example, river stages at Morrison Bridge during April–June could routinely reach +3.2 to +3.7 m in the 2080s, which is >2 m higher than what is typical in the recent period (Fig. 3). This upward shift in river stage reduced SWH in sections 1 and 2 by between 17% (+3.2-m stage) and 32% (+3.7-m stage) compared with available SWH estimated for recent period river stages (Fig. 3). A stage increase of 0.5 m as a consequence of climate change, from +3.2 to +3.7 m, nearly doubled the loss of SWH area in sections 1 and 2; these sections contained most of the SWH in the mainstem below Willamette Falls.

The proposed project's effect on SWH generally increased with river stage (Fig. 5). With a 470-m linear shoreline riprap footprint, the amount of SWH displaced by the proposed project increased slightly as river level increased (range 1.3–1.6% of the total SWH in section 1). Within the project footprint, at river stages estimated for 2080s flows, we estimated the project area contained >15% SWH that would be displaced by riprap (Fig. 5).

Spring Duration of SWH Extent

The duration of maximum SWH during subyearling peak occupancy varied by month and location. At Morrison Bridge (Rkm 20.6; which is representative of stages within river sections 1 and 2), April river stages during the recent period (1999–2009) encompassed the range for maximum SWH (+1.2 to +2.7 m) for 69% of the total number of April days during 1999–2009, and in May, river stages were in the peak range for 74% of the days (Table 1). Before 1937 there were fewer days in the river stage range of maximum SWH, 57% and 23% of days for April and May, respectively. Recent June river stages were in the maximum range for a longer period compared with historical June river stages (56% vs. 4% of days, respectively). However, historical June stages were typically not in the +1- to +2.7-m range because they were often much higher (Fig. 3). Historically, the lower river was connected to its floodplain, and river stages that regularly and substantially exceeded modern flood stage (5.5 m) represented additional but now lost opportunities for SWH formation throughout the study area. The duration of river stages near Willamette Falls (Rkm 42.2)

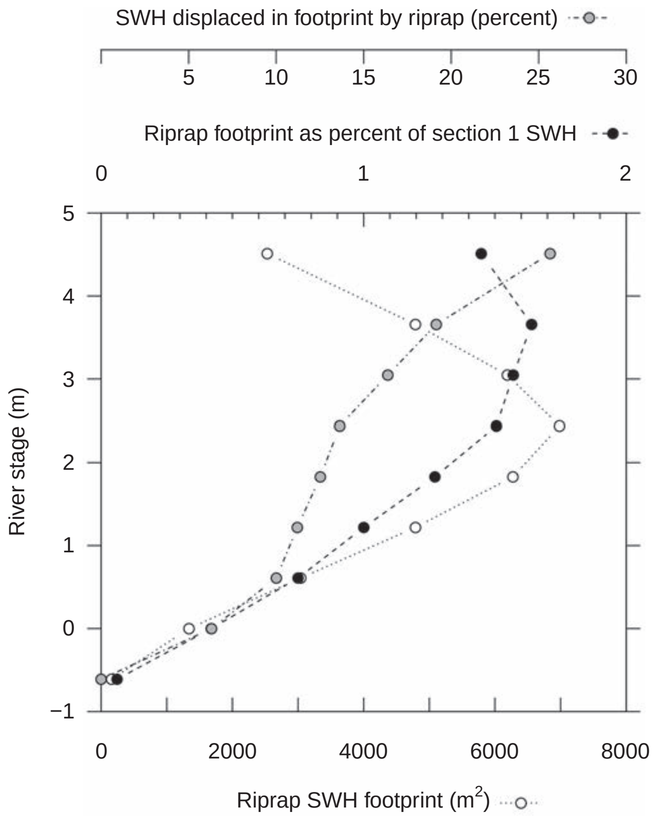

Figure 5. The effect of river stage on shallow water along low-sloped beaches (SWH) within the footprint of a hypothetical riprap installation in the mainstem Willamette River section 1 (Fig. 1) for 8 river stage levels at 2 spatial scales (percentage of the amount of SWH in river section 1 and at the project scale as a percentage of SWH displaced by the riprap).

Table 1. Estimated percentage of days during the spring (April–June) peak occupancy period of subyearlings in the mainstem that river stages were in the range that maximized the area of shallow water along low-sloped beaches (SWH) in the lower Willamette River at 2 locations.

	Days (%)		
Location[a] and period	*April*	*May*	*June*
Morrison Bridge			
1926—1936[b]	57	23	4
1999—2009	69	74	56
Below Willamette Falls			
1999—2009	33	36	47

[a] *Morrison Bridge Rkm 20.6, river stage range of maximum SWH: +1.2 to +2.7 m; below Willamette Falls, Rkm 42.2; river stage range of maximum SWH: +1.5 to +3.4 m.*

[b] *Historical stage heights at Morrison Bridge were much higher historically than the 1999-2009 period in June (Fig. 3); the number of days at stage heights of +1.2 to +2.7 m represented the lower end of the historical distribution of river stages typical for June. We are not aware of comparable historical river stage data below Willamette Falls.*

was less aligned with the peak level of SWH as a function of river stage (+1.5 to +3.4 m) (Table 1 & Fig. 5); more days in this range occurred during June than the previous 2 months of subyearling peak occupancy (Table 1).

Discussion

Our results suggest climate change is likely to have a large effect on the quantity of mainstem SWH, although different reaches have different proportions of SWH. The change in quantity also means the overall effect of a proposed riprap installation in SWH is different, and potentially more severe, than it would be if climate change effects were removed from consideration.

Sensitivity of SWH to river stages is particularly important in the Willamette River's lower river sections, which contain about 75% of the total SWH below Willamette Falls at current spring river stages. The area of SWH was substantially reduced relative to estimates of SWH area for the recent period by anticipated climate effects. Forecasts of Willamette and Columbia River flows (Figs. 2a-b) suggest that by the 2080s SWH in the lower river sections in April–June will decline as river stages increase (Fig. 3). The upper 2 sections showed the same general decrease in SWH, although they did not contain nearly as much SWH (Fig. 4). Estimates of river stages at Morrison Bridge, as a function of projected flows by the 2080s, showed that stages could be >2 m higher than recent levels during April–June, and at those higher levels we estimated that SWH in the lower 2 sections would be 17–32% lower than estimates for the recent period. Given these potential losses of SWH, a more detailed study that incorporates tidal effects, such as Kukulka and Jay's (2003) work in the mainstem Columbia River, and estimates of potential hydrogeomorphological changes shaping future SWH formation would more precisely quantify the action's and climate's effects on SWH.

Climate-induced increases in river stages estimated for the 2080s were similar to historical spring river stages (with unregulated flow) that resulted in seasonal annual flooding historically. Those opportunities for the river to expand into its floodplain no longer exist due to flow regulations and the presence of seawalls, dikes, and other flood-control measures. Most river systems have been subjected to a variety of anthropogenic alterations (Poff et al. 1997; Western 2001; Bunn & Arthington 2002) that affect dynamic habitat-forming processes, disconnect rivers from their floodplains, alter the composition of biological communities, and can block access to habitats, which may affect species survival and inhibit full expression of life-history variation (Bunn & Arthington 2002; Poff et al. 2007).

Assessing the effect of the riprap on SWH and its effects relative to climate conditions depended on the scale examined. Across the range of river stages we modeled, the project occupied an area between 1.3–1.6% of total SWH area in section 1 (Rkm 0–12.9), depending on river stage; the higher percentage occurred at anticipated 2080s flows. As the river stage increased so did the effect of the project. Within the project's footprint, about 15% of potential SWH was displaced by riprap at river stages corresponding to 2080s flows. Willamette River flows are managed according to guidelines established to achieve multiple objectives, including fish needs (NMFS 2008). Columbia River flows have a large effect on Willamette River lower mainstem stage dynamics. Thus, higher Columbia River spring flows by the 2080s, coupled with lower spring Willamette River flows as a consequence of reduced snowpack and earlier melting (Chang & Jones 2010), will have a large effect on stages during the peak abundance period for subyearling Chinook salmon. Currently, spring mainstem Willamette River SWH is not a factor considered in the management of flows for either river.

The mainstem Willamette River is more than a fish-migration corridor (Friesen et al. 2007). Salmonid life histories are complex and diverse, and use of the mainstem and estuary by salmon differ with the diversity of life-history types (Bottom et al. 2005; Teel et al. 2009). Regardless of whether the subyearling life history is an expression of phenotypic plasticity or has a genetic basis (Carlson & Seamons 2008), preservation of mainstem SWH would allow UWR Chinook salmon to express a larger suite of life-history strategies and would continue the potential for a subyearling emigrant life history. Maintaining life-history diversity is important for species persistence and recovery (McElhany et al. 2000), especially in the face of changing conditions, because it provides the raw material for evolutionary adaptation and allows populations to maintain themselves across a range of environmental conditions (Hilborn et al. 2003; Greene et al. 2009; Waples et al. 2009).

ESU Viability, Climate, and Habitat Alterations

Within-ESU diversity is used to characterize species' viability (McElhany et al. 2000; WLCTRT 2006; ODFW & NMFS 2011). Evaluations of UWR Chinook salmon viability have characterized this ESU as at high risk of extinction (ODFW & NMFS 2011). For many constituent populations of the ESU, mainstem habitat degradation has been implicated as one factor affecting juvenile survival and contributing to extinction risk. The importance of the subyearling life-history strategy to ESU viability probably varies from year to year and from population to population (Myers et al. 2006). However, significant reductions in habitats that support the spectrum of life-history strategies would further increase extinction risk. Mainstem habitat alteration and the compounded effects of climate and bank stabilization, such as we explored here, further decrease mainstem habitat.

The project's effect on the viability of UWR Chinook salmon, in concert with climate change, is hard to estimate because the importance of the subyearling life-history strategy to the ESU's overall resiliency and the relationship between subyearling survival and SWH is not known. However, given the preference of subyearling Chinook salmon for SWH we expect the effect to be detrimental. What is the magnitude of the project's effect? The scale of the project is small relative to the context of overall available habitats in the mainstem, so the effect would be small. What is the magnitude of the project's effect in the context of anticipated climate change? The effect increased across a range of increasing river stages. Given that the location of the project is on the shoreline, it would disproportionately affect a rare life-history strategy, making this strategy even more rare. Empirical studies directed at estimating UWR Chinook salmon subyearling survival in connection with mainstem habitat preferences, suitability, and use would increase understanding of the relationship between mainstem habitats and ESU viability.

Climate Change and Evaluation of Effects of Human Actions

Our case study demonstrated that effects of a human action on an ESA-listed species can be evaluated in concert with the effects of climate change, and that doing so will be an important component of fully understanding the effects of human actions on species. This will be especially pertinent when an action's effects can be expected to continue long into the future, such as the case with riprap. Including climate change in forward-looking population or habitat quality and quantity projections will be important in analyses supporting both ESA decision making and conservation decisions in general (McClure et al. 2013, this issue) to capture the full effect, whether positive or negative. Key components to build these projections include a clearly articulated link between climate, habitat, and the species' vital rates or habitat preferences; climate models at the appropriate scale and supporting secondary models, such as hydrologic predictions under climate change; and the biological effects of these conditions (Snover et al. 2013).

In situations where resource limitations preclude the ability to conduct a full modeling exercise, literature reviews may be helpful. Given the sheer number of small-scale projects that the ESA-consulting agencies contend with each year, we suggest that prioritizing geographic regions and project types for analysis of this sort may be important. Such full-scale analyses in one area can provide the touchstone for other consultations where a full modeling effort may be precluded. Results from the full effort could be interpreted within the context of the additional projects.

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Supporting Information

A description of the geospatial model and data links are available online (Appendix S1). The authors are solely responsible for the content and functionality of these materials. Queries (other than absence of the material) should be directed to the corresponding author.

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Bank Erosion as a Desirable Attribute of Rivers

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Bank erosion is integral to the functioning of river ecosystems. It is a geomorphic process that promotes riparian vegetation succession and creates dynamic habitats crucial for aquatic and riparian plants and animals. River managers and policymakers, however, generally regard bank erosion as a process to be halted or minimized in order to create landscape and economic stability. Here, we recognize bank erosion as a desirable attribute of rivers. Recent advances in our understanding of bank erosion processes and of associated ecological functions, as well as of the effects and failure of channel bank infrastructure for erosion control, suggest that alternatives to current management approaches are greatly needed. In this article, we develop a conceptual framework for alternatives that address bank erosion issues. The alternatives conserve riparian linkages at appropriate temporal and spatial scales, consider integral relationships between physical bank processes and ecological functions, and avoid secondary and cumulative effects that lead to the progressive channelization of rivers. By linking geomorphologic processes with ecological functions, we address the significance of channel bank erosion in sustainable river and watershed management.

Keywords: bank erosion, riparian ecology, fluvial geomorphology, sediment, aquatic ecology

Bank erosion is a natural geomorphic process or disturbance that occurs during or soon after floods. Riverbanks are transitional boundaries, or ecotones, between the aquatic and terrestrial ecosystems, and they frequently change under naturally dynamic hydrologic conditions. Although abundant evidence suggests that bank erosion is a necessary ecological process (Piegay et al. 1997, 2005), current river management, and sometimes even restoration strategies, calls for channel bank infrastructure, that is, hard structural elements intended to arrest bank erosion (also called revetment, erosion control, or bank stabilization structures). Such strategies often focus on human values that include property damage and land loss, flood hazards (Piegay et al. 1997, Casagli et al. 1999), and potential impacts to aquatic habitat from bank-derived fine sediment contributions (EPA 2007). Often, projects labeled as "restoration" focus principally on bank stabilization. However, static banks are not the norm, and static rivers and streams do not sustain ecosystems. Despite this, in response to the notion that bank erosion is deleterious, the construction of bank infrastructure has become pervasive over the past century as an increasing population and associated development encroach on riparian landscapes. Thus, bank erosion management is a significant ecological issue.

In this article, we review the ecological significance of a range of geomorphic bank erosion processes and show that the cumulative effect of progressive bank stabilization structures is to limit riparian function and diminish habitat for riparian species. Our objectives are to (a) synthesize geomorphic and biological literature through principles that highlight the importance of bank erosion processes as disturbances integral to components of riparian ecosystems at a variety of scales; (b) identify the effects of channel bank infrastructure on riverbank and riparian ecology; (c) identify failures of current policies to manage channel bank erosion; and (d) present a rationale and framework for alternatives to such policies. The alternatives are intended to aid the development of river management and policy that promote healthier geomorphological and ecological functions in river systems where bank erosion is an issue of concern.

Geomorphic and ecologic significance of banks and bank erosion

We define "riverbank," in a geomorphic context, as the landform distinguished by the topographic gradient from the bed of a channel along the lateral land-water margin up to the highest stage of flow or up to the topographic edge where water begins to spread laterally over the floodplain surface. Bank erosion refers to the erosion of sediment from this distinct landform. Eroded sediment moves along the topographic gradient laterally toward the channel or in the downstream direction. Banks are often characterized by bare sediment, live vegetation, or snags (Roy et al. 2003). In an ecological context, riverbanks are an important component of riparian zones. Bank habitat and function are to some degree inseparable from functions within the larger riparian zone; here we take a broader view of natural banks and bank erosion as they influence riparian areas. Ecologically

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functioning riparian zones provide a variety of resources and are vital centers of biodiversity (Gregory et al. 1991, Naiman et al. 1993, 2005, Ward and Tockner 2001, NRC 2002). The main functions of riparian zones are related to fluvial hydrology and sediment dynamics; retention and cycling of nutrients and pollutants; and maintenance of habitat for wildlife, including invertebrates, amphibians, reptiles, birds, and mammals (NRC 2002). In the following sections, we review elements of banks and bank erosion that create physical and biological heterogeneity and riparian diversity. We focus discussion of bank processes and functions around principles that illustrate the significance of bank erosion and natural banks as desirable attributes of rivers:

- Bank erosion provides a sediment source that creates riparian habitat.
- Active banks create and maintain diverse structure and habitat functions.
- Riparian vegetation promotes bank stability and contributes large woody debris.
- Bank erosion modulates changes in channel morphology and pattern.

Channel banks form a significant ecotone between aquatic and terrestrial ecosystems with diverse structure and habitat functions; this article forms the critical basis for discussions of the effects of and alternatives for channel bank infrastructure.

Bank erosion provides a sediment source that creates riparian habitat. Diverse bank erosion processes occur as sediment cycles through the continuum of headwater to lowland environments within a watershed (figure 1). The dominant bank erosion process in each part of the watershed is influenced by the size of the channel, discharge, and flow strength (Couper 2004), with the dynamic nature of erosion processes depending in part on sediment supply and transport regime (Benda et al. 2004). Fluvial deposits vary dynamically from the headwaters to the lowland (Church 2000). Bank erosion from headwater areas provides a source of weathered sediment that is stored for varying periods in downstream alluvial deposits (Gomi et al. 2002).

Bank erosion is a considerable sediment source in some rivers (Trimble 1997); however, the sediment supply is not always deleterious. Bank erosion supplies coarse sediment to channels—a size fraction that is necessary to form the physical structure of aquatic habitat. Coarse sediment, supplied from upstream and stored as channel-bed material and bedforms, makes up substrate important for macroinvertebrates. Such coarse-grained substrate promotes oxygen exchange, provides interstitial space for protection from predators, serves as attachment sites for filter feeders, and provides a food source for periphyton (Wood and Armitage 1997). In contrast, when the sediment supply is large relative to transport capacity, such that aquatic habitat is buried, or when fine-sediment contributions from bank erosion are excessive, habitat damage may occur. In streams with large sediment inputs derived from bank erosion, there is often concern that changes in water quality due to large fine-sediment loads affect aquatic habitat (EPA 2007). Large fine-sediment inputs may affect groundwater-surface water exchange, a factor in fish and benthic invertebrate habitat (Lisle 1989, Kondolf et al. 2006). Processes that include infiltration of fine-grained sediment into coarser channel substrate may in turn impede intergravel water flow in the hyporheic zone, consequently reducing oxygen levels to benthic organisms.

As a physical process that supplies and delivers sediment, bank erosion is critical for creating habitat at the watershed scale (figure 1). Riparian area structures are influenced by variations in geomorphic processes and in the resulting valley bottom deposits, including floodplains and bars (Gregory et al. 1991). Floodplain ecosystems, a critical component of riparian ecosystem diversity (Ward and Stanford 1995, Stanford et al. 1996), are sustained by periodic erosion and sedimentation during floods (Junk et al. 1989, Bayley 1991, 1995, Florsheim and Mount 2002). Bank erosion also contributes sediment to fluvial deposits, such as sandbars in the Platte River, that are important to migrating whooping cranes (*Grus americana*). Resting on the bars during their migration, these birds have long sight lines and are isolated from predators (NRC 2002, Graf 2005).

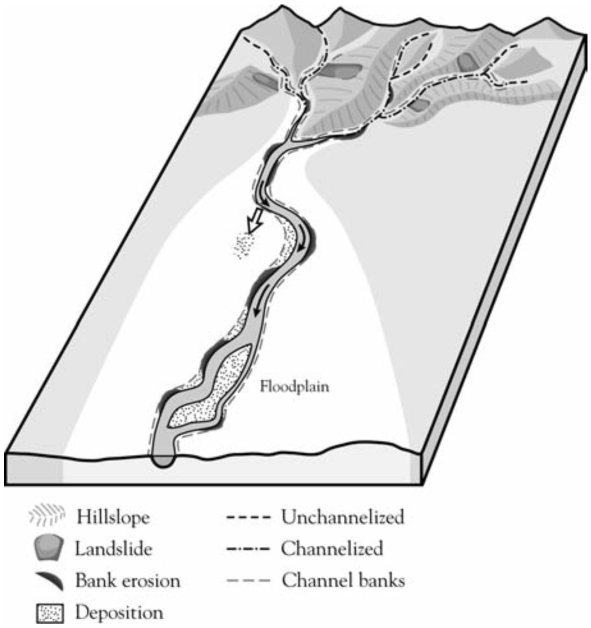

Figure 1. Illustration of a river network from headwater to lowlands. Bank erosion is one component of the sediment cycle throughout an idealized river network. In the headwaters of watersheds, banks are the boundary between upland terrestrial and aquatic ecosystems. In lowland areas, channel banks are commonly the transitional area between floodplain and aquatic habitats. Sediment eroded from hill slopes in headwater areas is transported downstream and stored in deposits (such as terraces, floodplains, bars, and channel substrate) that provide habitat for aquatic and riparian organisms.

Active banks create and maintain diverse natural structure and habitat functions. As a transitional zone within riparian ecotones, riverbanks accommodate highly dynamic environmental conditions. Banks can modulate floodwater surface elevations and have variable moisture regimes that satisfy the requirements of diverse plant species (NRC 2002). Banks provide habitat at different elevation zones needed by flora and associated fauna adapted to flood pulses rising along the bank (Junk et al. 1989). Habitats along the bank gradient are exposed to various flood frequencies, durations, and magnitudes (NRC 2002, Naiman et al. 2005). Thus, riparian plant communities closest to a channel are colonized by fast-growing, water-adapted sedges, rushes, grasses, herbs, and seedlings of shrubs and trees, whereas terrestrial vegetation is deterred because of frequent flooding (Gregory et al. 1991, NRC 2002). At higher elevations on the bank, riparian plant communities include trees such as cottonwood (*Populus*), willow (*Salix*), and alder (*Alnus*), whose roots are adapted to periodic floods (NRC 2002). Vines such as the riverbank grape (*Vitis riparia*) climb riparian trees, and wildlife consume their fruit. Streamside trees that overhang the channel are an allochthonous source of organic material that provides food and cover for fish. Additionally, organic material from riparian vegetation is a primary food source for invertebrates from all of the guilds, including filter feeders, shredders, scrapers, and predators (NRC 2002). Streamside trees offer shade that modifies aquatic microclimates and maintains lower water temperatures (NRC 2002). Bank erosion alters the gradient of vegetation during floods, and thus modifies the habitats and functions of the riparian ecosystem. Bank erosion that locally opens the tree canopy increases primary production and energy flow through the food web, leading to greater production of invertebrates and fish (Naiman and Bilby 2001).

The channel banks and vegetation within riparian areas make up the substrate for insects emerging from the water, and those insects provide a food source for breeding and migrating birds (Benke and Wallace 1990, Graf et al. 2002). Dense, newly established vegetation patches formed following erosional, depositonal, or flood disturbances offer habitat for diverse bird species (table 1).

Amphibians that require water for part of their life cycle, such as frogs, toads, and salamanders, rely on bank microhabitat for dispersal onto land after emerging from the water (NRC 2002). Many reptiles require functioning riparian areas to complete their life cycles. For example, the wood turtle (*Clemmys insculpta*) establishes nesting burrows in recently deposited, unconsolidated sediments of riparian areas (Vogt 1981, NRC 2002, Harding 1997). Snakes hunt in biologically rich riparian ecotones (NRC 2002). Riparian lizards (*Sceloporus occidentalis*) eat river-derived insects, which highlights the energy flux between rivers and surrounding

Table 1. Effects of channel bank infrastructure to control bank erosion.

Geomorphic and ecological attribute	Habitat or ecosystem service influenced	Examples of organisms affected
Loss of sediment source		
Supply	Downstream sandbars as resting habitat for migrating birds	Whooping crane (*Grus americana*)
Grain size	Coarse-grained substrate for attachment and interstitial space for hiding from predators	Macroinvertebrates (e.g., mayflies [*Ephemeroptera*], caddisflies [*Trichoptera*], and stoneflies [*Plecoptera*])
Loss of geomorphic processes		
Migration	Newly scoured or deposited surfaces	Riparian trees (e.g., cottonwood [*Populus*], willow [*Salix*], alder [*Alnus*])
Widening	Adjustment necessary for incised channel to evolve toward equilibrium with floodplain at elevation to support riparian plants	Riparian trees (see above)
Loss of bank substrate		
Unconsolidated sediment	Vertical banks for wildlife burrowing and nesting Filter and retention of nutrients, pollutants, water quality	Bank swallow (*Riparia riparia*) Macroinvetebrates (see above)
Natural biotic and abiotic components of land-water margin	Shoreline microhabitat: soft sediment or burrows, emergent vegetation to cling to; underwater plants, snags, roots protruding from bank	Shore-dwelling insects (e.g., *Neocurtilla*); macroinvertebrates
Roughness and irregularity in land-water margin	Variation in near-bank flow velocity, refugia during storm flows	Overwintering fish, macroinvetebrates (see above)
Undercut banks	Protection from predators	California shrimp (*Syncaris pacifica*), juvenile fish (e.g., Coho salmon [*Oncorhynchus kisutch*])
Loss of riparian forest		
Stream-side riparian ecosystem Willow and cottonwood forests	Complex riparian vegetation, areas for wildlife: bird breeding, nesting, safety from predators; probing for insects under tree bark; wildlife: food, migration corridor, and/or dispersal route; plants: structure for vines	Birds (e.g., willow flycatcher [*Empidonax traillii extimus*], Gila woodpecker [*Melanerpes uropygialis*], western yellow-billed cuckoo [*Coccyzus americanus occidentalis*]), reptiles (e.g., riparian lizard [*Sceloporus occidentalis*]), semiaquatic mammals (e.g., river otter [*Lontra canadensis*]), macroinvertebratres, climbing vines (e.g., river-bank grape [*Vitis riparia*])
Overhanging branches, leaves	Shade, organic material, fish food	Fish, macroinvetebrates (nymph and adult stages)
Large woody debris	Reduction in pool complexity and depth, loss of attachment sites	Fish, macroinvertebrates (see above)

terrestrial areas (Sabo and Power 2002). Semiaquatic mammals such as the water shrew (*Neomys fodiens*), star-nosed mole (*Condylura cristata*), beaver (*Castor*), river otter (*Lontra Canadensis*), and mink (*Mustela*) find food and shelter resources in riparian habitats (NRC 2002). Natural banks and associated vegetation offer cover for these animals while they move back and forth between water and land.

Riparian vegetation promotes bank stability and contributes large woody debris. Riparian vegetation influences bank stability (Simon and Collinson 2002) because the type and density of vegetation cover and the roots that stabilize banks minimize bank erosion (Pizzuto and Mecklenburg 1989, Abernethy and Rutherfurd 1998, 2000). Riparian forests generally maintain bank stability, but flow that scours around individual pieces of large wood derived from riparian forests may accelerate bank erosion rates locally—this contrast highlights the importance of considering scale in assessing bank erosion (Montgomery 1997). During floods, bank erosion delivers large woody debris to channels (Piegay et al. 1999, Wyzga and Zawiejska 2005, Sudduth and Meyer 2006). The large woody debris changes bed and bank morphology and increases channel complexity (Ralph et al. 1994). Pool formation in forested ecosystems is controlled in part by the size and abundance of large woody debris, but other factors are also important (e.g., sediment supply; Buffington et al. 2002). In rivers with fine substrate, large woody debris provides a stable substrate for organisms in channels otherwise dominated by highly mobile, fine-grained bed sediment (Junk et al. 1989).

Bank erosion modulates changes in channel morphology and pattern. Bank erosion includes two main processes that are often interrelated: mass wasting processes and fluvial erosion (Hooke 1979, Thorne 1982, Odgaard 1987, Osman and Thorne 1988, Thorne and Osman 1988, Hasegawa 1989, Lawler 1993, Darby and Thorne 1996, Lawler et al. 1997, ASCE 1998, Simon and Curini 1998, Casagli et al. 1999). Mass wasting processes on riverbanks include various types of slides (e.g., shallow or deep slides) and slab failure characterized by linear or rotational failure planes. Slides occur when the driving force exceeds the resisting force during floods or shortly after storm flows recede. Subsequent floods may erode sediment deposited in the channel from a slide. However, while the sediment remains at the base of the bank, it may locally increase the physical heterogeneity of the channel through the addition of large woody debris and cobbles, and the creation of microtopography and bare surfaces at various elevations above the channel bed.

Fluvial erosion occurs during floods when the near-bank flow velocity and acceleration exert shear stress on the banks that is greater than the critical shear stress needed to entrain bank sediment. Fluvial erosion frequently scours the toe of the bank, causing the upper portion to collapse (Thorne and Tovey 1981). The relation between the rate of sediment supply from bank erosion and the rate of fluvial transport of this material from the base of the bank controls the rate of bank retreat (Thorne 1982). Floods that cause erosion are stochastic, and local field conditions—as well as human modifications—are highly nonuniform. Thus, measurement and prediction of long-term erosion rates is complex; in practice, there are numerous challenges in extrapolating temporal and spatial scales of bank erosion (Couper 2004).

Fluvial erosion of bank sediment may expose tree roots or undercut and destabilize bank vegetation. Alternatively, if bank sediment bound by a root network resists erosion, flow may undercut banks below the roots, forming new niches for crustaceans, mollusks, or juvenile fish to hide from predators and find low velocity refugia during floods. For example, the California freshwater shrimp (*Syncaris pacifica*) prefers tree-lined banks with underwater vegetation, where it can rest on exposed roots in the summer and seek shelter by clinging to roots exposed in undercut banks during winter floods (Biosystems Analysis 1994). Additionally, fluvial erosion that scours sediment from the base of riverbanks maintains habitat for some avifauna, such as the bank swallow (*Riparia riparia*), which relies on unconsolidated bank sediment for nesting (Garrison et al. 1987). Erosion is a critical process in maintaining the vertical banks that preclude predators' access to bank swallow nests and prevent vegetation from covering the birds' habitat. Nesting colonies move to new sites along a river each year, taking advantage of new vertical banks that form following bank erosion.

Mass wasting and fluvial erosion at bends drives episodic or progressive channel migration and changes in channel pattern, which influence the establishment of riparian vegetation. Bank erosion is associated with long-term evolution of channel pattern and short-term geomorphic adjustments that alter morphology, including widening, migration, braiding, and avulsion and associated channel abandonment. Thus, the influence of bank vegetation on erosional resistance is a control that, along with other fluvial variables such as river slope and discharge, influences alluvial river patterns (Millar 2000). Bank erosion may occur on one or both banks in incising, aggrading, or in laterally migrating channels (ASCE 1998)—adjustments that lead to the formation of new scoured surfaces. Thus, bank erosion provides new niches for vegetation requiring sunlight and lack of competition. Recruitment of woody plant species such as cottonwood and willow occur on such alluvial surfaces (NRC 2002).

Bank erosion is especially prevalent, and erosion rates are highest, on the outside of river bends, where fluvial processes, mass wasting, and undercutting of riparian vegetation leads to meandering (e.g., Leopold and Wolman 1957, Johannesson and Parker 1989, Hupp and Osterkamp 1996). Bank erosion that facilitates meandering and creation of abandoned channels is important because it leads to vegetation succession, which is necessary for riparian diversity (Salo et al. 1986). Riparian plant succession is initiated with the establishment of patches of seedlings that favor bare substrate created during floods (Friedman and Auble 2000). As point bars and vertically accreted sediment deposits extend, and younger vegetation becomes established after subsequent floods, vegetation patches increase in age in a direction opposite to the migrating channel (Everitt 1968, Naiman et al. 2005).

Bank erosion also occurs in relatively straight, braided, or multiple-channel systems, and is often associated with changes in water and sediment supply that lead to incision (Simon et al. 1999, Thorne 1999). Channel adjustments that increase bank height and instability in incised channels ultimately lead to widening and deposition of sediment surfaces at elevations that support the establishment of riparian trees (Simon 1989). In braided channels, bar accretion may lead to local bank erosion when the flow is diverted around a bar toward the bank. In multiple channel systems, bank erosion facilitates avulsion, which creates new channel habitat patches within the floodplain and leaves others abandoned. Thus, bank erosion is one component of an array of geomorphic processes that govern channel evolution and lead to the morphologic diversity in habitat needed to sustain riparian biodiversity.

Effects of channel bank infrastructure

Channel bank infrastructure such as riprap, gabions, or concrete lining is increasingly common in agricultural, rural, and urbanizing areas, where its usual purpose is to limit land loss and associated hazards and damages. Many types of hard material are used (figure 2). Structures vary in extent from the scale of the individual bank erosion feature to longer reaches associated with urbanization or flood control projects that are kilometers long. Table 1 identifies and summarizes the main geomorphic and ecological effects of channel bank infrastructure, the potential habitat or ecosystem services lost, and examples of organisms affected.

Hard bank structures increase flood velocities along banks, preventing the establishment or survival of many riparian plant species (NRC 2002); thus, bank stabilization can have negative effects on riparian areas (Sedell and Beschta 1991, Fischenich 1997). Channel complexity tends to be reduced by the changes that channel bank infrastructure produces: elimination of bank irregularity and channel-width variations, homogenization of near-bank flow velocity, loss of access to side channels, loss of natural bank substrate, and limitation of geomorphic adjustments. Moreover, complex riparian areas offer a greater variety of food sources and physical habitats than do simple plant communities of uniform age and species, which are characteristic of stabilized banks (Gregory et al. 1991). Completely arresting bank erosion disrupts the lateral channel-bank sediment exchanges that are necessary to sustain an array of aquatic habitats (table 1).

Figure 2. Examples of channel bank structures. Some bank erosion-control structures are not designed or engineered; rather, they are ad hoc attempts to prevent local land loss or damage. (a) Car bodies; (b) riprap; (c) sacrete on left bank, riprap on right bank; and (d) rock-filled gabions along banks of concrete-lined channel. Photographs: Joan L. Florsheim (2a and 2c) and Anne Chin (2b and 2d).

Land-use changes that remove riparian vegetation have a significant influence on channel banks (Allan 2004). Hard erosion-control structures eliminate substrate for and microhabitats of plant species that grow on banks. They also impede the movement of species that use riparian zones for migration corridors, reduce structural integrity offered by roots, destroy reptile nesting areas, and diminish habitat for avifauna (NRC 2002). For example, willow habitat for the southwestern willow fly catcher (*Empidonax traillii extimus*) is threatened on the heavily modified Rio Grande in Colorado and on other southwestern rivers in the United States (Graf et al. 2002). Similarly, unconsolidated bank substrate habitat for the bank swallow is destroyed by riprap.

Removal of riparian vegetation reduces shade and energy input from fallen leaves, and can raise stream water temperature and primary production (Quinn 2000). Loss of riparian vegetation also reduces the volume of wood in channels (Johnson et al. 2003). Habitat created by large wood in channels once provided essential overwintering habitat, but is now considered a key limiting factor for coho salmon and other fishes in the Pacific Northwest (Moyle 2002). Similarly, deforestation of tropical ecosystems limits wood availability to pools, which plays a role in structuring fish communities and increases aquatic diversity (Wright and Flecker 2004).

In streams where riparian vegetation is removed from banks to make way for erosion control structures, it follows that macroinvertebrate production, essential for aquatic food webs, is often diminished. The diversity and density of aquatic macroinvertebrates are higher in streams with wider riparian areas (Newbold et al. 1980). Roy and colleagues (2003) found the strongest relationships between various macroinvertebrate indices and forest cover within a 100-meter-wide riparian buffer zone. The ecological consequences of erosion control infrastructure in urbanizing rivers include the removal of vegetation and the loss of habitat for macroinvertebrates (Sudduth and Meyer 2006).

The use of erosion control structures that reduce deleterious effects on biota has advanced in the past few decades (Downs and Gregory 2004, Chin and Gregory 2005). Recent engineering approaches often incorporate vegetation in the structure design to reduce habitat degradation. Despite inclusion of large woody debris or living vegetation in some channel bank infrastructure, however, two important geomorphic issues arise: (1) channel bank infrastructure fundamentally alters geomorphic processes, and (2) structures may be ineffective, especially over the long term. Gilvear (2000) noted that bank erosion-control structures might fail when flood magnitudes exceed the discharges for which the structures are designed, or when processes such as channel migration are ignored. Because hard structures, even when they incorporate vegetation, impede geomorphic adjustment processes, they can lead to more damaging erosion events locally or in downstream reaches (Henderson 1986, Arnaud-Fassetta et al. 2005). Nevertheless, bank erosion-control structures can be effective in minimizing land loss over decadal timescales (Shields et al. 1995), although some evidence suggests that they are ineffective over multidecadal timescales and potentially have secondary effects (Larsen and Greco 2002, Thompson 2002). Thus, the geomorphic and ecological effects of channel bank infrastructure may be severe, although generally little monitoring is done to assess the effects or the effectiveness of projects that use channel bank infrastructure (Kondolf and Micheli 1995, Harris et al. 2005). As a management strategy, construction of channel bank infrastructure addresses only one component (bank erosion) of the full spectrum of habitat degradation and environmental problems found in developing watersheds—problems such as channel incision, removal of riparian vegetation, changes in hydrology, and pollution (Booth 2005, Meyer et al. 2005).

Shortcomings of current riverbank management

The causes of bank erosion are complex and often combine disparate geomorphic processes, such as fluvial erosion and mass wasting. However, riverbank stabilization structures often are designed to address only fluvial erosion, and thus fail on banks where mass wasting processes are predominant (figure 3).

Failure to understand bank erosion processes and functions. Fluvial erosion and mass wasting processes both lead to channel migration, a mechanism that maintains the ecological structure of riparian ecosystems (Bravard and Gilvear 1996) and the width adjustments necessary for river morphology to adapt to incision and episodic or variable sediment loads. Thus, bank erosion is integral to sediment transfer, river evolution, and ecosystem sustainability. In fact, bank erosion is a necessary process that may bring about eventual channel stability in urbanizing systems (Chin 2006). Henshaw and Booth (2000) suggested that construction of channel bank infrastructure should not be an immediate response in watersheds with a low level of urban development or where development is in progress, because hard structures may prevent the adjustments required for a channel to stabilize on its own. Further, Sudduth and Meyer (2006) suggested that total elimination of bank erosion should not be a goal of habitat restoration because limiting bank erosion simplifies complex natural channel morphology. Thus, the short-term benefits of bank erosion-control infrastructure on geomorphic processes and ecological function may come with relatively high long-term environmental costs.

Failure to consider bank erosion management at the appropriate scale. Channel bank infrastructure constructed at the local scale is often implemented structure by structure over the short term by individual landowners or by government or public agencies. Such practices do not consider bank erosion in the geomorphic or ecological context of the appropriate temporal and spatial scales—namely, long-term and systemwide scales.

Couper (2004) pointed out the importance of defining and linking scales because rates of erosion measured over the course of long-term river evolution contrast with rates documented for a short period of time within a channel reach. Various river resource and regulatory agency management guidelines (Flosi et al. 1998, McCullah and Gray 2005, EPA 2007) address bank erosion processes at the scale of an identified erosion site even though channel bank erosion is a river management issue best addressed at the watershed or ecosystem scale. Rarely is the spatial extent or temporal frequency of bank erosion processes documented in the comprehensive manner necessary for long-term, watershed system–scale analyses. Moreover, the potential effects of global warming on geomorphic processes (Tucker and Slingerland 1997, Goudie 2006) are rarely considered in bank erosion management. Failure to consider the spatial distribution, extent, and temporal frequency of both bank erosion and bank erosion-control infrastructure at the scale of the watershed over the long term precludes understanding of the influence of bank erosion processes on both geomorphic and ecological functions. Without considering these scales, understanding the secondary and cumulative effects of bank infrastructure is not possible.

For example, bank erosion is a critical concern within California's Sacramento River system because eroding stream banks threaten levee integrity. The US Fish and Wildlife Service (USFWS 2000) estimates that more than half of the river's banks on the lower 310 kilometers of the Sacramento River were riprapped during the past 40 years as part of the Sacramento River Bank Protection Project. Governor Schwarzenegger brought the erosion issue to the policy forefront in the 2006 declaration of a state of emergency for California's levee system. The emergency declaration directed the California Department of Water Resources (DWR) to identify and repair erosion sites in the state-federal project levee system "in order to prevent catastrophic flooding and loss of life." More than 100 erosion sites were documented along the main stem (excluding tributaries) of the Sacramento River in 2005, and more than 20 were reported as critical, with bank erosion progressively threatening levee integrity. In 2006, DWR and the US Army Corps of Engineers undertook 21 levee repairs on the river's main stem (DWR 2006). Maintaining the dynamic Sacramento River in response to episodic erosion mechanisms carries a great economic and environmental cost—in particular to the bank swallow—and as a river management approach, it is not currently sustainable. Nor will the system be sustainable in the future, should flood discharges in the Central Valley increase, as they are predicted to do as a result of climate change (Dettinger et al. 2004).

Failure to consider secondary effects. Channel bank infrastructure that limits the geomorphic processes that transfer sediment through dynamic natural systems may lead to undesirable secondary effects. For example, such structures may reduce sediment supply to channels. In addition, such structures can shift the locus of erosion as the river adjusts to the hardened area that the structure presents. Bank structures can narrow channel width, leading to higher flow strength and thus initiating a cycle in which the increased flow strength, in combination with reduced sediment supply, leads to channel deepening. The deepening may in turn increase bank height and accelerate bank erosion. Thus, in deepening channels, bank structures may become ineffective and may be destabilized by continuing erosion.

Figure 3. Failure of sacrete bank erosion control structure because of high pore-water pressure in the bank behind the structure. Photograph: Joan L. Florsheim.

Failure to consider long-term and cumulative effects. In many fluvial systems, hard bank erosion-control structures already exist, products of previous erosion control efforts. Over time, these structures are joined by new ones erected to armor new erosion sites, producing assorted generations and styles of channel bank infrastructure, all within short reaches of the same channel. As each new structure interacts with geomorphic processes, bank erosion may shift to a new location, creating a chain reaction in which each new section of eroded bank is armored with new erosion control structures. One consequence of channel bank infrastructure that has long-term effects (beyond the design life of the structure) is that a structure may preclude future restoration attempts designed to incorporate self-design and self-sustaining habitats (figure 4). If cumulative long-term effects are not taken into consideration, the result could be progressive construction of channel bank infrastructure that, although intended to limit local bank erosion, tends toward eventual channelization of entire river systems.

Alternatives to channel bank infrastructure

Alternatives to channel bank infrastructure that provide a vision for sustainable river management must accommodate dynamic geomorphic processes that sustain ecological functions and habitat on channel banks. Figure 5 identifies management actions and alternatives necessary to accommodate bank erosion processes. These actions are intended to reverse past and current failures in riverbank management. First, it is imperative to understand bank erosion processes and functions in diverse riparian systems. This requires the identification and assessment of geomorphic

Figure 4. Bank erosion processes continuing behind large rock riprap originally placed at the base of the bank. If left isolated in the channel, riprap may become an impediment to future restoration. Photograph: Joan L. Florsheim.

processes, ecological functions, and the likely effects of channel bank infrastructure. Second, it is imperative to consider bank erosion management at the appropriate temporal and spatial scales—that is, at the watershed scale over the long term, even if the extent of local erosion is small. Doing so will help avoid treating the symptom rather than the cause of erosion. Third, the secondary effects of any approach to modulate erosion must not interfere with the potential for future restoration initiatives or with the natural river adjustments needed to maintain equilibrium. Finally, to conserve aquatic and terrestrial riparian habitat, long-term and cumulative ecological and geomorphic effects must be considered in the context of the legacy of past and potential future projects.The four alternative approaches discussed below provide a conceptual framework to help planners and policymakers address bank erosion issues (figure 5).

Dynamic-process conservation areas are defined here as zones with sufficient area to accommodate bank erosion along with other dynamic processes, such as flooding. This approach accommodates geomorphic processes active within a watershed's sediment transport system over the long-term instead of focusing on the local scale, at which processes are episodic and erosion is transient. Designation of the appropriate extent of dynamic-process conservation areas could be accomplished through integrated ecological and geomorphic scenarios for restoration. Process-based restoration (Wohl et al. 2005) promotes floodplain functions, such as flooding, and inclusion of secondary channels, floodplain lakes, or marshes that rely on connectivity (Buijse et al. 2002). Dynamic-process conservation areas support connectivity and conservation of habitat and services needed by organisms that utilize riparian areas (see table 1). This alternative could be achieved through the development of long-term strategies to acquire riparian and adjacent land, land-use planning within riparian-centric governance structures, and multiagency and private or nongovernmental organization partnerships.

An *erosion easement* is a legally binding restriction placed on private or public riparian land to allow bank erosion processes to operate. Easements to accommodate geomorphic processes and ecological functions could be a component of a riparian buffer that promotes habitat or ecosystem services (see table 1). Designating the appropriate extent of an erosion easement depends on a thorough assessment of bank erosion processes and fluvial system evolution at the watershed scale; Piegay and colleagues (2005) addressed methods of quantifying appropriate widths on the basis of geomorphic processes. As with strategies to develop dynamic-process conservation areas, implementation of this alternative would require long-term land-use planning in order to purchase land and obtain landowner agreements along both riverbanks within riparian corridors.

Elimination of direct stressors, the impacts caused by human activities or land uses that directly cause or accelerate bank erosion processes, is a relatively simple way to enhance bank stability. For example, grazing is a stressor that leads to riparian vegetation denudation; however, the impact may be eliminated through exclusionary fencing, which keeps cattle from damaging stream banks and riparian vegetation in rangeland. This option could be implemented in concert with all the other alternatives to decelerate bank erosion through land-use planning, best-management practice guidelines, or ordinances.

Nonstructural approaches are those that do not contain hard elements such as large rocks, concrete blocks, root wads, or large woody debris as construction materials. Such approaches

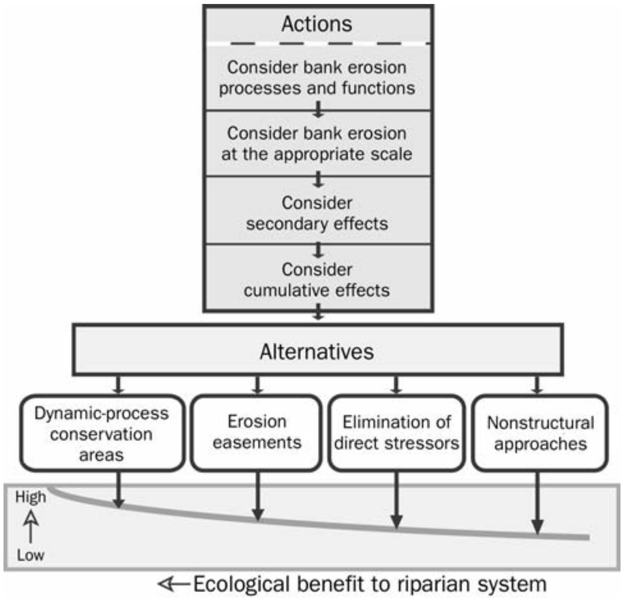

Figure 5. Framework for alternatives to channel bank infrastructure. Dynamic-process conservation areas protect the linkage between river channels and adjacent landscapes, and provide the highest ecological benefit to riparian ecosystems. The other alternatives provide ecological benefits to the degree that they accommodate the geomorphic processes that sustain them.

include planting native vegetation without inclusion of hard elements. In particular, willow sprigs are commonly planted to promote root networks that bolster bank strength. Fences are sometimes constructed of willow branches, which later take root and sprout. Such alternatives may not completely arrest bank erosion, but they may be beneficial when the management aim is short-term moderation of erosion processes that does not inhibit the potential for future restoration or preclude the long-term benefits of alternative management approaches such as dynamic-process conservation areas or erosion easements.

Transcending traditional notions of bank erosion management

Pervasive construction of infrastructure to control bank erosion—a product of the notion that bank erosion is deleterious—has greatly diminished natural channel banks, geomorphic processes, and ecology. Management approaches that aim to arrest bank erosion at the scale of the transient erosion site are spatially constricted and consider only the short term. Hard structures may include vegetation, but they cannot sustain or restore riparian functions in urban or rural areas. Thus, the challenge is to develop sustainable bank management alternatives that preserve aquatic organisms and riparian plants, birds, and other wildlife (see table 1).

Differentiating between extensive or chronic bank erosion caused by human activities and land uses versus those caused by natural geomorphic processes and river evolution warrants attention in current science and management efforts. In order to protect riparian functions, river management and policy decisionmakers must determine when channel bank infrastructure is warranted on the basis of societal needs. Management decisions to implement channel bank infrastructure may be necessary in some cases to protect public safety; however, an appropriate starting point for discussion is science-based policy that promotes conservation and restoration of river processes and channel bank habitat and functions. Policy based on alternatives illustrated in figure 5 stems from a growing understanding that bank erosion is one geomorphic process inexorably linked with ecological functions. Global river management efforts (Brookes 1995, Kauffman et al. 1997, Piegay et al. 1997, Cals et al. 1998, Gilvear 2000, Golet et al. 2003, Palmer et al. 2005, F. Nakamura et al. 2006, K. Nakamura et al. 2006) and research that promotes conservation and restoration of natural processes support the alternatives presented in this article.

Conclusions

Bank erosion is one component of the natural disturbance regime of river systems and is integral to long-term geomorphic evolution of fluvial systems and to ecological sustainability. Bank erosion is therefore a desirable attribute of rivers. Four shortcomings in current river management are the (1) failure to understand and accommodate bank erosion processes and functions, (2) failure to consider bank erosion management at the appropriate scale, (3) failure to consider secondary effects of bank erosion-control infrastructure, and (4) failure to consider long-term and cumulative effects of bank erosion-control infrastructure. These failures are often synergetic. For example, rarely is the spatial extent or temporal frequency of bank erosion processes documented comprehensively enough to allow for long-term watershed-scale analyses that could illuminate the cumulative effects of channel bank infrastructure. Such analysis is necessary to avoid the progressive channelization of rivers. To address current and past management failures, we identify and discuss broad alternatives to accommodate geomorphic processes that promote riparian functions: (a) dynamic-process conservation areas, (b) erosion easements, (c) elimination of direct stressors, and (d) nonstructural approaches, such as those that include live vegetation that may moderate bank erosion processes without limiting long-term geomorphic evolution. Combining bank management goals that conserve diverse natural bank habitat and riparian vegetation with policies that accommodate erosion processes and watershed-scale sediment cycling and river evolution contributes to a strong basis for sustainable river management.

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The benefits of expanding riverbank habitat in the Russian river watershed

Investing in our quality of life

Produced by Russian Riverkeeper
July 2014

Author's Note

This report does not intend to lay blame on any particular party but instead to point out stressors that prevent our full attainment of the services that the ecosystem along the banks of the Russian river provides. We include this information to help the reader understand how combined forces of environmental and societal change are cumulatively reducing the health of the river. Our ultimate goal in writing this report is to spread awareness of the value of riverbank habitats.

As we sat down to begin writing this report we were quickly reminded of the dynamism of the fields of river conservation and restoration. New opinions and realizations continued to surface during the production of this material. We recognize that the opinions and data below may be updated after publication of this report and welcome any questions, concerns, comments or suggestions that readers may have.

We intend for this report to be accessible to a diverse range of audiences including local farmers, citizens, business people and regulators. The content contains both the best available science and our suggestions for how this information might be utilized to benefit the river's health in a cost-effective and efficient manner. River protection is a collaborative affair. It is essential that we all work together to permanently protect the health of this beautiful and essential component of our local economy.

1.0 Introduction

If you have ever visited or made your home in Sonoma or Mendocino county, the Russian river likely influenced your decision. Winding its way down through the Laughlin range in

Mendocino county, continuing its journey south to Sonoma county, and eventually reaching its end at the Pacific Ocean in Jenner, the river plays a central role in the development of the economy and charm of the region.

Our region's remarkably successful wine industry, a central driver of the growth of local prosperity, draws its strength from the valuable climate and soils that the river provides. Local agricultural entrepreneurs have been transforming these 'natural products' into more readily consumable wines for generations. Russian explorers and settlers brought the European wine-making tradition to Sonoma County in the 1830s and the resulting industry has brought wealth and global recognition to our beautiful home ever since.

Unfortunately, our continued usage and dependence upon the river has also had a negative impact upon its health. Many coexisting human influences combine to produce a cumulative threat to the river's ability to continue sustaining our quality of life. Local wildlife are all affected. Many native species of birds, fish, reptiles, amphibians, and mammals depend on riverbank vegetation as habitat. Recent declines in wildlife populations are attributed to loss of habitat.

While the main stem of the river was once left free to meander across the valley floors through which it flows, it is now constrained by competing and perpetually higher priority human uses of the land. Urban and agricultural development of land that once was the territory of the wandering river has squeezed its flow into the confines of modern engineering. Unable to unleash its energy upon lands adjacent to its main stem, known as *floodplains*, the river must direct its energy elsewhere. This phenomenon results in ecologically unsound and unsafe erosion of channel banks and deepening of the river bed. Experts refer to these processes as *incision* and *entrenchment*, respectively. These phenomena negatively impact the security of our local freshwater supply.

The many competing users of the river's flows and the water diversions that satisfy them compound the negative impact of the river's physical restrictions. Not only does the river have no place to exercise its innate drive to wander and change, but it also is increasingly unable to maintain the flows necessary to provide its vital services to our community.

One particularly important sector of our local economy, the wine industry, requires a reliable water supply from the Russian river for both irrigation and frost control. The health of this industry is vitally important to the health of our regional economy and is also dependent on an increasingly vulnerable and waning resource, sufficient flows in the Russian river.

Incision and entrenchment has lead to reduced frequency of groundwater recharge by natural over-bank flooding and lowering of the water table. Loss of wildlife populations due to habitat loss is resulting in increased regulatory compliance costs as more species fall under the protection of state and federal environmental regulations.

The river's vital role as a source of drinking water and place of recreation is consistently threatened by pollution. Polluted urban and rural storm water, wastewater discharges and agricultural runoff transport the various waste products of our economy directly to the waterway from which we drink and in which we swim. Pollution threatens native plants and wildlife as well as human livelihoods. It also poses a substantial cost to the community to remediate.

We can only wait and see how these concerns will be impacted by global climate change. Less frequent but more intense storms will force us to reevaluate traditional strategies to quench our societal thirst for the water and other *ecosystem services* that only a healthy and functional Russian river can provide. The U.S. National Drought Mitigation Center classified 76.4% of California as experiencing ‘extreme drought’ as of the completion of this report (U.S. Drought Mitigation Center). Our region will need to implement new strategies to improve our resilience against the future impacts of climate change and preserve the security of the local natural resources upon which we depend.

2.0 What’s Wrong with Existing Riverbank Habitat?

2.1 Background

The current state of riverbank habitat along the Russian river is inadequate to sustain its ecosystem services. The cumulative impact of various consequences of human development have vastly reduced the extent of native riverbank vegetation and negatively altered river processes that provide ecosystem services. Existing habitat is fragmented and often too narrow to provide adequate functionality (Hilty 2004). The loss of habitat in our watershed reflects the greater trend in California of exchanging loss of riparian ecosystem services for more houses, shopping centers, vineyards, roads and other forms of development. About 90% of riverbank habitat in California has now been lost (Point Blue 2014). In the Russian river watershed alone, 34 percent of riverbank habitat between Healdsburg and Wohler bridge has been lost since 1942 (San Francisco Chronicle 1995).

This loss of habitat also reflects a similar regional trend of habitat loss. The Western United States experienced a net loss of 374,600 acres of wetland habitat between 1982 and 1987, which was 5.7 percent of the total wetland area of the region at the time. Much of this habitat loss included conversion of riverbank habitat to other uses. Some of these uses include agricultural diversion, water diversions, construction of levees and flood control structures, construction of dams and reservoirs, excessive livestock grazing, and urbanization (CA Department of Fish and Wildlife 2014). Riverbank habitat loss is a national environmental issue of concern, but it represents a particular threat in California as there is so little original habitat left.

2.2 Incision, entrenchment and cascading loss of riverbank habitat

The causes of the loss of riverbank habitat in the Russian river watershed are complex. The human uses of riverbank habitat listed above have not only led to direct conversion of habitat but have also set off a series of geomorphic processes that negatively impact the river’s provision of all ecosystem services. These processes are incision and entrenchment.

Historical mining of gravel deposits drastically reduced natural sediments supply in the river. When a river is deprived of sediments, it gains more erosive force and begins to scour its bed. When this occurs, most of the erosive force of the river is redirected from natural over bank flooding into adjacent floodplains to scouring its bed. This results in a deepening channel bed and a shift from a wide U-shaped river profile to a more rectangular profile with steeper banks. This process is known as incision while the resulting rectangular, deepened state of the river is known as entrenchment.

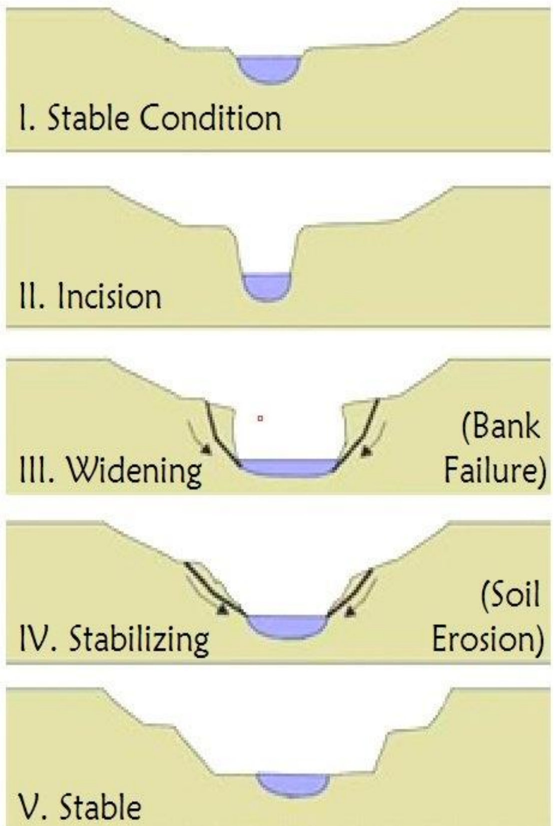

Figure 1. Graphic depicting a river cross-section as it succumbs to incision and subsequently adapts. (2003, USGS)

Incision usually results in a narrower channel that provides less aquatic habitat, reduces riverbank habitat, and provides less bed area for infiltration of river water for groundwater recharge (Darby and Simon 1999). As the channel narrows, the river applies additional force to its banks as it attempts to return to its previous wider profile. As the channel widens, bank failure is likely to occur, as depicted in stage III of the graphic in Figure 1. You can picture that as the channel deepens and then river banks fail that any floodplain level riparian vegetation or former backwater channel is left high and dry. Once a channel incises it also draws down the water table with the river bed elevation and dries out the much higher current floodplain. Once people see the trees like cottonwoods die the response is to cut them down, which often leads to removal of any other remnant vegetation and loss of large woody debris from the river.

This channel widening is the current source of many bank failure incidents in the main channel of the Russian river. Unfortunately, bank stabilization projects, natural or otherwise, are currently the preferred solution to these incidents. Restricting further widening of the riverbanks through rip-rap or bank stabilization with rock only halts the rivers' attempts to correct the channel's incised state. This strategy only stalls the inevitable widening process and it does not prevent it. Accordingly, when the river repeats its attempt to correct its form, perhaps aided by storm-induced floodwaters, the bank stabilization projects are subjected to extensive damage. Bank stabilization projects tend to lock in a narrow inadequate riverbank buffer that limits future growth of riparian vegetation. Since they goal is to stabilize the banks it eliminates the river's ability to create habitat such as cut-off meanders, cut banks and pool/riffle complexes that create habitat for Salmon.

If we are to prevent dangerous bank failures from occurring we must allow them to occur. This may seem paradoxical but it is actually a means to work WITH natural processes as opposed to against them in attempting to restore the functioning of the interconnected channel and floodplain system. The only means to permanently halt all erosion is to line the entire Russian river with concrete like the La River and obviously no one wants that to happen. The alternative is endless and costly bank projects that continue to erode the river's ability to provide water purification, flood control or provide enough wildlife habitat to keep species from going extinct.

3.0 Why Protect the Russian River and its Banks?

3.1 <u>Background</u>

The Russian river watershed contains a unique and incredibly important set of ecosystems that collectively furnish the thriving economy and quality of life that local residents have come to expect from their home. The most important ecosystem lies along the banks of the main channel and its tributaries. Riverbank vegetation provides many services to wildlife and people. The river provides drinking water to approximately 600,000 residents of Sonoma, Mendocino and northern Marin counties (Sonoma County Water Agency). In addition, it contributes to our region's flourishing tourism industry by attracting thousands of visitors who come to swim, fish and paddle the highly navigable waters. Neither of these gifts could be provided without the support of riverbank vegetation.

There is a growing trend among researchers, policy makers and business people to define ecosystems in a manner that is more useful within policy contexts. Conflicts between nature and industry are often inhibited by the reality that it is difficult to describe the value of nature in the language of business. To date, important questions such as, how much money do vineyard owners save due to the flood protection services of Russian river riverbank habitat, go unanswered. Working towards answering these types of questions will enable decision makers to more effectively defend the value of protecting nature from human development.

In order to begin answering such questions, researchers have developed a new term to define the value that society derives from nature. This term is known as ecosystem service. "Ecosystem services are the benefits provided by ecosystems that contribute to making human life both possible and worth living” (UK National Ecosystem Assessment). The term is further broken down into four types of services, provisioning, regulating, supporting and cultural. Provisioning services include the products derived from nature such as wood or fresh water. Regulating services are obtained from the regulation of ecosystem processes such as riparian habitat’s ability to filter pollution out of agricultural runoff. Supporting services are the baseline services that are necessary for the production of all other ecosystem services. These include natural processes such as nutrient cycling and soil formation. Cultural services refer to non-material benefits that humans receive from ecosystems such as recreational and tourism opportunities.

Unfortunately, the river’s ability to continue providing these and many more essential services to our region is threatened. The dual concerns of global climate change and increasing human development are drastically inhibiting the river’s ability to clean our dirty water, reduce the intensity of floods, provide habitat for plants and wildlife, and provide water to irrigate local agriculture and quench our thirst.

All of these ecosystem services that are provided to our region depend on the existence of healthy riverbank vegetation. The Environmental Protection Agency affirmed the importance of this habitat when it explained to water utilities that, “Protecting, acquiring, and managing ecosystems in buffer zones along rivers, lakes, reservoirs, and coasts can be cost-effective measures for flood control and water quality management” (EPA Climate Ready Water Utilities 2012).

3.2 <u>Benefits for wine grape growers</u>

Healthy riverbank habitat provides the following ecosystem services to vineyards adjacent to the Russian river:

- Flood peak attenuation and storage
- Increased groundwater recharge
- Decreased regulatory compliance costs
- Erosion reduction over time
- Decreased regulatory cost from species extinction

In 1995 the Russian river watershed experienced some of the worst floods in years. Locals bore witness to excessive damage to public infrastructure such as roads and power transmission lines as well as to many vineyards planted adjacent to the river.

The San Francisco Chronicle ran a special report on the disaster. The article quotes a CA Coastal Conservancy staffer’s observations, “‘In the Alexander Valley where they had ripped out the trees along the river, all of the debris coming downstream washed out vineyards and caused more damage than in those places where the trees were left to buffer the agriculture’” (1995). These vineyards suffered more flood damage than necessary because they were located in former riverbank habitat located in a floodplain that succumbs to periodic floods.

Furthermore vineyard development in recent years has removed many trees just outside the river channel that provide a second line of defense against erosion when riverside trees erode. When there is only 1-2 tree widths between the river and a vineyard or other land use once that one line is lost there is nothing left to protect against erosional forces.

Undisturbed riverbank habitat depends on regular flooding to sustain it biodiversity and ecological functionality. In fact, many native riverbank species depend on natural flood events to sustain part or all of their life cycle. The Fremont cottonwood tree requires high floods to create moist sediment bars where its seeds may germinate (Darby and Simon 1999). Native riverbank vegetation has evolved to thrive in highly flood-prone environments. A secondary effect of these species' honed durability is that they provide excellent protection against the very floods that power their life cycles. The California Department of Fish and Wildlife recently released a memorandum summarizing the ecosystem services provided by riverbank habitat. The report states expanding habitat "to encompass the geomorphic floodplain is likewise desirable to optimize flood-reduction benefits" (CA Department of Fish and Wildlife 2014).

Vineyard owners that allot a healthy amount of land for riverbank habitat adjacent to their vines will not only experience less damage when inevitable flood events occur, but will also benefit from renewed groundwater supply as attenuated floodwaters slowly seep into the floodplain soil.

In addition to the value of increasing healthy riverbank habitat area as a defensive measure against flooding, there is also value in healthy habitat's pollution reduction and habitat provision functions. Both of these functions may reduce a landowner's potential compliance costs related to the California and federal endangered species acts. California Department of Fish and Wildlife affirms this statement, stating that "[riverbank] restoration can have a major influence on achieving the goals of the Clean Water Act, the Endangered Species Act, and flood damage control programs" (2014).

The Clean Water Act requires most development that has an adverse impact on water quality objectives of waterways such as the Russian river to mitigate their impact.

Although there are currently no requirements for vineyard establishments, the federal and state Clean Water Acts are currently developing regulations to control water pollutants from vineyards as part of the Agricultural Lands Discharge Program. Ultimately all vineyards will need to demonstrate that they meet water quality standards intended to protect the Russian river's beneficial uses of water such as habitat for salmon and other wildlife, flood peak attenuation and storage, wildlife migration corridors, and municipal water supply. The final regulation will have to meet stringent cleanup rules since the river is listed as polluted for fine sediment and agricultural cultivation is a primary source of this pollutant.

One simple method for reducing sediment delivery to streams is to increase vegetated riverbank area that filters sediment and other vineyard pollutants. Wider riverbank vegetation would meet regulatory requirements and provide greater benefits than current vineyard best management practices for water quality including improving wildlife habitat, reducing flood impacts, decreasing future regulatory compliance costs and better protection of vines from eroding channel banks.

3.3 Benefits for the greater community

Healthy riverbank habitat provides the following ecosystem services to the general public in Sonoma and Mendocino County:

- Flood peak attenuation
- Increased rate of groundwater recharge
- Recreation
- Pollution reduction (reduced costs)
- Reduction in endangered species listings

Historical floods have causes excessive damage to public infrastructure. As climate change continues to contribute to more intense flood peaks in future storms, we can expect to incur even greater damage to critical infrastructure located in floodplains such as roads and homes. It would be prudent to limit future development in floodplains and to implement wider habitat setbacks so that native vegetation may once again provide a defense against flood waters. Native riverbank vegetation excels at slowing flood waters down and encouraging them to soak into the ground rather continue flowing over the surface. An added bonus to this benefit is that sinking floodwaters often replenish precious groundwater supplies that might be the only water source in prolonged future droughts.

Thousands of people visit the Russian river every year to swim, fish, paddle, and relax on its beautiful beaches. Healthy riverbanks help keep the river clean so that swimmers and fishing enthusiasts don’t come away from their recreational pursuits with irritating rashes or a side of mercury poisoning with their fish dinner. Local businesses that are regularly frequented by tourists benefit from the mass appeal that the river has among visitors. The river is a critically important component of Sonoma and Mendocino county’s reputation and brand and thereby drives significant flows of capital to local business.

Healthy riverbanks are highly effective at filtering polluted water. If they are well maintained, they may be more cost effective than traditional forms of water treatment at addressing several known pollutants. There is an extensive body of research demonstrating that riverbank vegetation stores sediment and retains and transforms excess nutrients, pesticides and other toxic substances (Riley 2009). Riverbank vegetation is particularly effective at removing sediment from agricultural runoff. Studies have shown that this vegetation can remove 80-90 percent of sediment in agricultural runoff.

A water treatment plant costs much more to build and operate than a riverbank restoration and maintenance program, yet riverbanks can provide equivalent pollution reduction service. Riley estimates that a recently built plant in Santa Monica provides equivalent water treatment as 4,000-5,000 lineal feet of riverbank habitat. While it costs about $1.3 million ($2008) per year to run the plant, a typical restoration project costs $227,000 per year

($2008). Even a large multi-objective flood damage reduction project costs just $967,600 per year ($2008). Investing in sound riverbank and floodplain restoration for pollution filtration is a cheaper alternative constructing traditional water treatment infrastructure. Communities can reduce water infrastructure fees by investing in the filtration capacity of healthy riverbank vegetation.

3.4 Benefits for wildlife

Healthy riverbank habitat provides the following ecosystem services to wildlife with life cycles that rely upon riverbank vegetation:

- Shelter
- Food supply
- Water temperature regulation
- Pollution reduction

California Department of Fish and Wildlife has stated that the best available science concludes that a habitat width of at least 164 feet is necessary to maintain viable habitat for many of California’s wetland dependent native species, including birds, reptiles, and amphibians (2014). Current riverbank setback policy in the watershed only protects 50 feet of habitat, measured from the edge of bank to upland edge of habitat. More riverbank habitat is needed to preserve native species populations and to avoid increases in habitat mitigation costs resulting from development in listed species habitat.

Table 1. Currently listed species that depend on Russian river riverbank habitat. (County of Sonoma).

Species	Listing	Type of Organism	Habitat Requirements
Western yellow-billed cuckoo	CA ESA, Endangered	Bird	Historical occurrences riparian woodland and scrub.
California freshwater shrimp	US ESA, Endangered	Shellfish	Riparian scrub and woodland in perennial drainages with undercut banks and overhanging vegetation.
Central California Coho salmon	US ESA, Threatened; CA ESA, Endangered	Fish	Juvenile and adult migrations occur in the spring and fall/winter, respectively. Juveniles of this species rear in small tributaries.

Species	Listing	Type of Organism	Habitat Requirements
Central California Coast Steelhead	US ESA, Threatened; CA ESA, Endangered	Fish	Juveniles emigrate primarily March through mid June, and adults migrate primarily from December through March. Although juvenile steelhead primarily rear in tributaries, they do occupy portions of the main stem Russian river.
CA Coastal Chinook salmon	US ESA, Threatened	Fish	Juveniles emigrate primarily March through June, and adults migrate September through December (primarily late October through mid-November. Juvenile Chinook salmon migrate to the ocean shortly after hatching and do not rear in the main stem Russian river. Juvenile emigration is essentially completed by the end of June.

Habitat loss is one of the leading causes of observed declines in these species populations (CA Department of Fish and Wildlife 2014). Salmonids have suffered more than any of the other 67 fish species that are native to California. Coho salmon abundance has declined by at least 70 percent since the 1960s. Researchers predict that 78 percent of native California salmonids will disappear from the state within the next century if current population trends persist (2014).

While the task of reversing declines in native salmonid populations may seem daunting, there are many other species considered to be highly vulnerable to outside pressures. These plants and animals, known officially as species of special concern (SSC), may be of much greater significance to local landowners because declines in their populations can be readily reversed if riverbank habitat is expanded now. Twenty nine percent of native inland fish species in California fall under this category.

Native “SSC” species, or “species of special concern” are likely to be listed in the near future if riverbank habitat loss continued unabated. Every additional species added to the state or federal endangered list drives regulatory compliance costs skyward. Sensible landowners stand to reduce this financial risk by adopting management practices that encourage expansion of habitat. The CA Department of Fish and Wildlife has stated that it is often too late to recover native species once they are listed (2014). For landowners, this statement means that current and potential future costs of compliance with endangered species regulations will permeate their pocket books as long as the policies stand.

Every native amphibian in the Pacific Northwest depends on riverbank habitat during its lifecycle (CA Department of Fish and Wildlife 2014). Several local amphibians and reptiles are SSC, including the California red-legged frog, foothill yellow-legged frog, and Western pond turtle. If landowners act now to preserve and expand habitat, they stand to avoid future listing of these species and associated compliance costs.

Table 2. Vulnerable species that depend on Russian river riverbank habitat and likely to be listed in the future (CA Department of Fish and Wildlife 2014 and United States 2012).

Species	Listing	Type of Organism	Habitat Requirements
CA red-legged frog	US ESA, Threatened; CA ESA, SSC	Frog	Streams ponds, and marshes with permanent or temporary water bordered by emergent or riparian vegetation. Requires 4- 6 months of permanent water for larval development.
Foothill yellow-legged frog	CA ESA, SSC	Frog	Moderate to high gradient streams with gravel to cobble substrate. Breeds in pools with slower moving water.
Western Pond Turtle (also known as Pacific Pond Turtle)	CA ESA, SSC	Turtle	Slack or slow- moving aquatic habitat with available aerial and aquatic basking sites. Upland breeding sites are typically on unshaded, south facing riverbank slopes with soils of high clay or silt composition.
Russian river tule perch	CA ESA, SSC	Fish	Tule perch are abundant in the Russian river. Tule perch prefer pool habitats, and are known to inhabit the river immediately below the Mirabel rubber dam.

Species	Listing	Type of Organism	Habitat Requirements
California roach	CA ESA, SSC	Fish	Roach inhabit a wide variety of habitats in the Russian river Basin, but appear to be most abundant in small tributaries.

3.5 Benefits for all stakeholders threatened by climate change

Wildlife and humans will be negatively impacted by climate change within the next century. Our state's current drought is likely a sign of things to come later this century. Wang et. al found In 2009, the CA Natural Resources Agency released a statewide climate adaptation strategy that states that drought conditions "are likely to become more frequent and persistent over the 21st century due to climate change" (2009). Increasing frequency of drought conditions will make threaten native species and possibly result in more listing of species. If more species are listed as endangered, otherwise unremarkable drought conditions will have a greater impact upon grape growers and other farmers. More species listings will lead to greater water allocations for wildlife amidst stagnant or decreasing total water supplies.

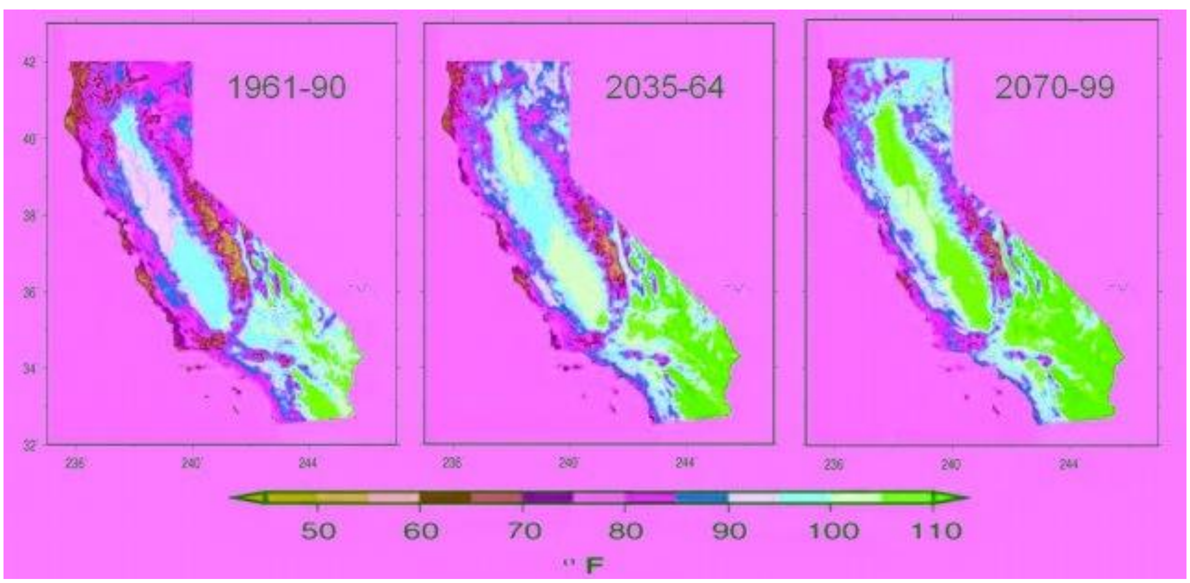

Figure 2. California historical and projected July temperature increase 1961-2009. (CA Climate Adaptation Strategy 2009).

The hydrological severity of the drought in 2009 was unremarkable when compared with similar historical droughts yet its impacts upon the San Francisco Bay Delta led to the first ever proclamation of a statewide drought emergency. Even if the severity of droughts does not change, their increasing frequency will exacerbate the impact upon farmers' water supplies, making it harder to find enough water for crops and afford regulatory compliance costs.

Climate change will also lead to increased frequency of extreme rainfall and flooding events. Local riverside landowners understand the potential financial consequences of inadequate preparation for floods. The damages incurred by the 1995 Russian river floods is a testament of potential future damages. These floods demonstrated the defensive capabilities of riverbank vegetation as riverside vineyards with intact riverbank forest incurred less flood damages than their counterparts without trees to block debris.

It makes economic and environmental sense to allocate more land area for riverbank vegetation now rather than later.

Lord Stern, the world's leading expert on the financial implications of climate change, has consistently stated that addressing climate now will be much cheaper than waiting until later. The key step that all farmers, particularly grape growers, who are dependent on riverine water must take is to allocate capital to strengthen their climate change resiliency. Allowing more room for riverbank vegetation to grow is effectively a natural insurance policy against climate change. Greater riverbank habitat area equals stronger protection against floods, more secure groundwater supplies, reduced likelihood of increased compliance costs as less species are added to endangered lists and decreased financial vulnerability to natural erosion events.

Greater riverbank habitat area will also benefit community members that rely on the river for drinking water and recreation and associated floodplains for protection against the brunt of flooding events. Grape growers stand to benefit both themselves and the greater community by investing in riverbank habitat. Here we could have a great example of grape growers' adoption of pro-habitat planting decisions improving livelihoods across the entire watershed.

Table 3. Distribution of riverbank ecosystem services among stakeholder groups.

We're talking about externalities that actually provide positive value for people and wildlife. In an era when the only externalities that are generally discussed are those spouting from factory exhaust vents, designating additional land for riverbank habitat is a particularly attractive investment for grape growers, other private landowners, the general public.

4.0 Pathways Forward

4.1 <u>The wider the better</u>

Wide riverbank habitat provides better services than narrow habitat. Studies have demonstrated a positive correlation between native wildlife abundance and riverbank habitat width (Hilty 2004 and CA Department of Fish and Wildlife 2014). Wider strips of native vegetation from more room for more native organisms. Narrow habitat strips may harbor more nonnative species including highly damaging domestic predators like dogs and cats. Cats, in particular, are known to prey upon native bird species (CA Department of Fish and Wildlife 2014). While the ideal habitat width varies greatly among different native species and site conditions, a good rule of thumb calls for riverbank habitat that is approximately 100 meters wide, including 50 meters of undeveloped upland habitat.

Current riverbank habitat setbacks simply do not provide enough land area for native species

Stakeholders	**Riverbank Ecosystem Services**
Wildlife	shelter, food supply, temperature regulation, pollution reduction
Greater human community	flood attenuation, recreation, filtration of drinking water and wastewater (at reduced cost)
Riverbank landowners (particularly grape growers)	flood attenuation, recreation, filtration of drinking water and wastewater (at reduced cost), groundwater recharge, decreased frequency of bank failure

to thrive. Creating a voluntary system by which individual landowners could allocate wider setbacks would help ensure that additional native species do not end up on government endangered lists.

Wider riverbank habitat can also filter more pollution out of runoff than narrow habitat (CA Department of Fish and Wildlife 2014). Wider habitat has more plant biomass above ground and below ground that captures pollutants like pesticides and fine sediments. More effective pollutant removal leads to healthier habitat, which leads to healthier native species that are less likely to drive up landowners’ compliance costs by becoming listed. More effective pollutant removal by wider habitat also reduces the financial burden upon the community for supporting traditional water treatment processes.

4.2 Address the source, not the symptom of the problem

Today, implementing many small scale bank stabilization projects seems to be the preferred river restoration strategy. The number of river restoration projects in the United States has grown rapidly over the past 2 decades. There were 100 known projects during the 1980’s. In 2001, there were over 4,000 projects (Lennox 2007). In that same year California ranked 3rd in stream restoration efforts, spending $3,699,785 per 1,000 miles of stream. Many of these projects consisted of planting trees to stabilize riverbanks. A significant portion of the funding for these projects comes from government agencies tasked with protecting water quality and endangered species habitat.

Considering that these agencies are by and large funded by our taxes, community members should be concerned with reducing restoration costs. Bank stabilization projects install natural vegetation or manmade infrastructure to reduce bank failure. While manmade infrastructure projects provide little to no value to the river system, natural vegetation bank stabilization projects at least provide limited habitat renewal. However, these projects are fragile and are often damaged or destroyed entirely by high flood events. When damage occurs, more funds must be expended to repair or replace them.

As climate change increases the intensity of flood peaks, repair costs for these projects will reach even higher levels. There's a reason why California's Climate Adaptation Strategy suggests funding large scale flood plain restoration projects. These projects act as effective sinks for flood waters, protecting riverside landowners and the greater community from potentially devastating flood damage and requiring less repair than other forms of restoration.

Communities can save money on excessive small scale restoration projects like bank stabilization by funding larger and more resilient projects such as floodplain restoration. These projects correct the damage incurred by decades of unwise land use within floodplains.

The first step to restoring floodplains is to assign wider riverbank habitat setbacks so that erosion events do not threaten landowners and do not require immediate repair. This strategy could potentially save taxpayers large sums of money.

While current Sonoma County General Plan requirements stipulate that agricultural development adjacent to the river's main stem allocate a 25-foot buffer from the top of bank for riverbank habitat, these mitigation measures do not completely eradicate project's contribution of sediment and other pollutants to the river.

The California and Federal endangered species acts require compliance measures that may incur substantial costs. These fees increase over time. WRA, a San Rafael-based environmental consulting firm described a recent fee increase in October 2013, "0.3 acre of wetland impact would have cost $2,162 [prior to fee increase], but now would cost $2751". They also state that Regional Water Quality Control Board fees tend to increase every two years. Increasing compliance costs due to poor environmental condition are an unnecessary financial risk to vineyard owners. As more riverbank habitat is converted to other uses, more species will become listed as threatened or endangered. As more species become listed, compliance costs will surge. These cost increases may be avoided by investing in greater land area allotments for riparian habitat.

Allocating more land area for riparian habitat will improve its functioning, allowing it to support the continued health of the many local species that depend on it during their life cycles. More habitat will lead to more shelter and more food for native species. This will reduce the likelihood of additional species becoming listed under the state or federal endangered species acts. This would provide a great benefit in regulatory cost avoidance especially if amphibians or reptile species that depend on both terrestrial and aquatic habitat are listed as endangered.

While the purpose of this report is not to dole out detailed solutions to habitat loss, we will note that there is plenty of precedent for sound solutions. The CA Natural Resource agency calls for the creation of floodplain corridors, which are effectively riverbank habitat setbacks that are larger than current requirements in our watershed. Government funds for floodplain conservation easements already exist. These easements have already been implemented in other parts of the state and country. Conservation easements coupled with restoration are an excellent potential strategy for expanding riverbank habitat.

There are many other potential solutions that are discussed in the associated companion report. Now that you have a clearer idea of what riverbank habitat loss means for our community, we invite you to reach out to discuss your thoughts on potential strategies to reverse historical losses.

5.0 Conclusion

Riverbank habitat in the Russian river watershed has declined drastically. The causes of local habitat loss are complex and interconnected. Conversion for urban and agricultural development is one of the main causes of declines. Current and past riverbank restoration projects have often focused on bank stabilization, which is rarely cost effective. Bank stabilization projects can require costly repairs and inadequately address natural erosion processes that sustain the health of the river. These projects are intended to halt erosion of banks. This strategy ignores the possibility that incision is an intermediary step in the river's natural method of healing itself.

Landowners should take the initiative to protect and expand existing riverbank habitat in order to preserve and enhance its ecosystem services. Wildlife, the greater human community, and riverbank landowners all stand to benefit from such actions. Benefits include reduced spending on regulatory compliance, pollution remediation, greater defense against climate change impacts and healthier habitat. Investing in the longer-term vision of pro-habitat, climate-friendly land use decisions is a smart financial, environmental and moral decision.

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TECHNICAL MEMORANDUM

DEVELOPMENT, LAND USE, AND CLIMATE CHANGE IMPACTS ON WETLAND AND RIPARIAN HABITATS – A SUMMARY OF SCIENTIFICALLY SUPPORTED CONSERVATION STRATEGIES, MITIGATION MEASURES, AND BEST MANAGEMENT PRACTICES

CALIFORNIA DEPARTMENT OF FISH AND WILDLIFE
NORTHERN REGION

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INTRODUCTION AND DOCUMENT PURPOSE

California Department of Fish and Wildlife (Department) Northern Region staff has analyzed the most recent and best available scientific research on the essential relationships of fish and wildlife to wetland, stream and riparian habitats, the impacts of land use and development on these habitats, and potentially effective conservation strategies to minimize these impacts. This technical memorandum is a summary of this analysis and has three principal objectives: 1) present a scientific analysis of fish and wildlife habitat needs and potential development and land use impacts, 2) detail potential conservation strategies and mitigation measures that have been effective in minimizing these impacts, and 3) make this scientific analysis available in the Northern Region to project proponents, consulting engineers and biologists, planners, California Environmental Quality Act (CEQA) lead agencies, the public, and to Department staff to inform project and land use plan design and review subject to CEQA. The Department's Northern Region serves Del Norte, Humboldt, Lassen, Mendocino, Modoc, Shasta, Siskiyou, Tehama and Trinity counties.

This Technical Memorandum also reviews relevant potential impacts of climate change and sea level rise on the Northern Region's wetland and riparian habitats. Over the current century, climate change will alter the fundamental character, production, and distribution of the ecosystems upon which the economy of California relies (Snyder et al. 2002, Snyder and Sloan 2005, California Energy Commission 2009a, 2009c). This climate change and sea level rise analysis is intended to inform Department staff and the public of these impacts as they relate to wetland and riparian habitat conservation and future local and regional land use and development decisions. This analysis does not address land use and development-related greenhouse gas emissions or their effect on climate change.

This document is intended to be a resource during the Department's participation in CEQA project review and land use planning in the Northern Region and to assist local agencies and the public during land use planning and development permitting processes. However, the Department affirms that project-specific circumstances always necessitate project-specific analysis of impacts and mitigation measure efficacy.

This summary of scientific literature is provided as a tool to be used, where appropriate, to support site specific project review and is not intended to be relied upon absent, or in lieu of, a site or project-specific analysis of environmental impacts and mitigation measures.

This technical memorandum includes the following:

1) Review of the Department's conservation and management role and legal authority;
2) Definitions;
3) Discussion of the historic loss and degradation of wetland and riparian habitats;
4) Review of the importance of these habitats and some of the species assemblages that depend upon them;
5) Assessment of potential development and land use impacts on these habitats;
6) Evaluation of projected climate change impacts for northern California;
7) Review of habitat buffer effectiveness;
8) Key findings that summarize effective mitigations and conservation strategies;
9) Commonly used methods for implementing wetland and riparian habitat buffers;
10) References that comprise the scientific basis of this Technical Memorandum.

DEPARTMENT ROLE AND LEGAL AUTHORITIES

Under Fish and Game Code section 711.7, the Department is designated as trustee for the State's fish and wildlife resources. The Department has jurisdiction over the conservation, protection, and management of fish, wildlife, native plants, and habitat necessary for biologically sustainable populations of those species (Fish & G. Code, § 1802). The Department administers the California Endangered Species Act (CESA) and other provisions of the Fish and Game Code that conserve the State's fish and wildlife public trust resources.

California lawmakers have identified a public interest in protecting and maintaining the State's wetland and riparian habitats. (Fish & G. Code, §§ 1385, 2780). In 1993, Executive Order W-59-93 established a comprehensive wetlands policy for the State that sought no overall net loss and long-term net gain in the quantity, quality and permanence of wetlands acreage and values. The Fish and Game Commission also has adopted a non-regulatory Wetlands Resources Policy, which recognizes the habitat values of wetlands and the damage to fish and wildlife resources from projects resulting from net loss of wetland acreage or habitat values (Fish and Game Commission 2013a). The policy, available at: http://www.fgc.ca.gov/policy/ and most recently amended in 2005, states:

> *"…it is the policy of the Fish and Game Commission to seek to provide for the protection, preservation, restoration, enhancement and expansion of wetland habitat in California.*
>
> *Further, it is the policy of the Fish and Game Commission to strongly discourage development in or conversion of wetlands. It opposes, consistent*

with its legal authority, any development or conversion which would result in a reduction of wetland acreage or wetland habitat values. To that end, the Commission opposes wetland development proposals unless, at a minimum, project mitigation assures there will be "no net loss" of either wetland habitat values or acreage.

The Commission strongly prefers mitigation which would achieve expansion of wetland acreage and enhancement of wetland habitat values."

The Department is a trustee agency pursuant to CEQA and also frequently serves as a responsible agency (Pub. Resources Code §§ 21069, 21070). The Department's role in wetland protection is primarily advisory in nature. The Department fills this role by reviewing and commenting on lead agencies' environmental documents and making recommendations to avoid, minimize, and mitigate potential negative impacts to those resources held in trust for the people of California.

WETLAND AND RIPARIAN DEFINITIONS

According to the U.S. Fish and Wildlife Service classification of wetlands (Cowardin et al. 1979), wetlands include swamps; freshwater, brackish water, and saltwater marshes; bogs; vernal pools; periodically inundated saltflats; intertidal mudflats; wet meadows; wet pastures; springs and seeps; portions of lakes, ponds, rivers and streams; and all other areas which are periodically or permanently covered by shallow water; or dominated by hydrophytic vegetation, or in which the soils are predominantly hydric in nature. Pursuant to the Fish and Game Commission Wetlands Resources Policy, the Department utilizes the U.S. Fish and Wildlife Service wetlands definition for purposes of wetland identification. The U.S. Fish and Wildlife Service wetlands definition is (Cowardin et al. 1979):

"Wetlands are lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water. For the purposes of this classification wetlands must have one or more of the following three attributes: (1) at least periodically, the land supports predominantly hydrophytes; (2) the substrate is predominantly undrained hydric soil; and (3) the substrate is nonsoil and is saturated with water or covered by shallow water at some time during the growing season of each year."

The State Water Resources Control Board (SWRCB), through a Technical Advisory Team, has also developed a working definition for wetlands, though this definition is not yet formally adopted (SWRCB 2012a):

"An area is wetland if, under normal circumstances, (1) the area has continuous or recurrent saturation of the upper substrate caused by groundwater or shallow surface water or both; (2) the duration of such saturation is sufficient to cause anaerobic conditions in the upper substrate

and; (3) the area either lacks vegetation or the vegetation is dominated by hydrophytes."

The SWRCB has also developed a working definition for riparian areas, which is based in part on Brinson et al. (2002) (SWRCB 2012b):

"Riparian Areas are areas through which surface and subsurface hydrology interconnect aquatic areas and connect them with their adjacent uplands (Brinson et al. 2002). They are distinguished by gradients in biophysical conditions, ecological processes, and biota. They can include wetlands, aquatic support areas, and portions of uplands that significantly influence the conditions or processes of aquatic areas."

HABITAT LOSS AND DEGRADATION

Temperate freshwater wetlands are threatened globally by urbanization, agriculture, hydrologic modification, and other land use practices and continued reductions in wetland area and function is likely to continue over the coming decades (Brinson and Malvarez 2002). On a national, state-wide, and regional scale, wetland and riparian habitats have undergone substantial declines. Over the past 200 years, the contiguous 48 states have lost an estimated 53 percent of their original wetlands, with California losing the largest percentage (91 percent) (Dahl 1990). An estimated 93 to 98 percent of California's and 75 percent of the North Coast's riparian habitat has been converted to other land uses (Katibah 1984, Dawdy 1989). On a local scale, salt marsh habitat in Humboldt Bay, California's second largest estuary has been reduced by 85 to 90 percent since 1897, due to diking and filling (Barnhart et al. 1992).

California and the nation continue to lose wetland acreage and value, despite both state and national regulations and "no net loss" wetland policies (National Research Council 2001). Between 1982 and 1987, the western United States experienced a net loss of 151,600 hectares (374,600 acres) of wetland habitat, a 5.7-percent loss in total wetland area (Brady and Flather 1994). Of the 67 aquatic habitat types in the Sierra Nevada, nearly two-thirds are in decline (California Department of Fish and Game 2007). Reasons for this loss are numerous and include: agricultural conversion, water diversions, construction of levees and flood control structures, dam and reservoir construction, excessive livestock grazing, and urbanization (Abell et al. 2000, Zedler 2004). According to Brady and Flather (1994), urban, industrial, and residential development was the greatest human-induced cause of wetland loss from 1982-1987.

According to the U.S. Army Corps of Engineers (USACE) and U.S. Environmental Protection Agency (USEPA), wetland creation and the restoration and enhancement of existing wetlands are a common means to mitigate for wetland loss (see USEPA, USACE 2008). However, on average, the quality of created, restored, and enhanced wetlands achieved through mitigation is lower than that of intact, reference wetlands, according to SWRCB-funded studies (Ambrose and Lee 2004, Ambrose et al. 2006). This suggests that projects conducted in wetlands, as currently permitted, are contributing to a net loss of wetland functions and values. According to Zedler (2004), mandatory compensatory wetland mitigation measures continue to result in a net loss of wetland habitat and the cumulative effects of historical and future wetland degradation will be difficult to abate.

An analysis of 45 Washington State compensatory wetland mitigation projects required pursuant to the Clean Water Act showed only 29 percent were implemented according to plan and also met the project's ecological performance standards (Johnson et al. 2000). This study also found that of 23 compensatory mitigations actually implemented, only 45 percent were implemented according to plan. Numerous studies have shown that wetland mitigation projects often do not meet their required USACE permit conditions (Kihslinger 2008). Along with the risk of mitigation underperformance or failure, the temporal loss of wetland function from the time of impact to the time a mitigation site is fully functional is also a factor in potentially diminishing the value of compensatory restored wetlands (Zedler 2004). Such temporal loss may vary depending on habitat type and other factors. For the above reasons, the Department, the California Coastal Commission, and others have often recommended mitigation for the loss of high-quality wetlands and riparian habitat at creation-to-loss ratios of 3:1 or greater.

Today, almost all of California's major rivers are dammed and diverted to provide water for agriculture and domestic use and many are channelized and constrained by levees to provide flood control for the farms and cities that now occupy land that was once seasonally flooded wetland and riparian habitats (Mount 1995, Moyle 2002, California Department of Fish and Game 2007, Mac et al. 2008). Maintaining the hydrologic connectivity of these floodplain habitats with the surrounding landscape, even if the habitats themselves are already protected from development, is critical to maintaining their biological integrity and ecosystem functions (Pringle 2001, Correll 2005, Tockner et al. 2008). California's wetland and riparian habitats are often part of an integrated ecosystem. California has 251,000 acres of riverine wetlands, approximately 9 percent of the state's total wetland acreage, and these wetlands are associated with 410,000 miles of rivers and streams (Snow 2010).

Isolating a river's floodplain from overbank flows degrades riparian habitat (Poff et al. 1997, Reckendorfer et al. 2013). When levees, berms, and canals disconnect rivers from their natural floodplains, they change the river's natural flow regime and eliminate the benefits of natural flooding such as deposition of river silts on valley-floor soils and the recharging of wetlands (Poff et al. 1997, California Department of Fish and Game 2007). In addition, disconnecting natural floodplains simplifies riverine and riparian habitat and diminishes braided channel structure and off-channel backwater areas, thus degrading habitat suitability for salmonid fishes (Moyle 2002, California Department of Fish and Game 2007, Tockner et al. 2008).

Non-structural approaches to floodplain management, such as not rebuilding flood-damaged structures in flood-prone areas and moving people out of harm's way are congruent with floodplain and riparian habitat restoration (National Research Council

1992, Sparks 1995). According to the Interagency Flood Management Review Committee (1994), the nation should discourage new development in floodplains as a means to prevent future flood damages and to help restore ecosystem function. Furthermore, global threats to human water security and to river biodiversity are well correlated Vörösmarty et al. (2010), as exemplified by enduring conflicts over water use and protection of declining species in the Klamath River Basin and Sacramento/San Joaquin River systems. Thus efforts to restore and protect riverine ecosystems, including floodplains and riparian habitat, will likely benefit both biodiversity and California's needs for a safe and reliable water supply.

As described in detail below, many California fish and wildlife populations that rely on wetland and riparian habitat, e.g. willow flycatcher (*Empidonax traillii*), western red bat (*Lasiurus blossevillii*), and coho salmon (*Oncorhynchus kisutch*), have plummeted in recent decades due, in part, to habitat loss and degradation. Wetland and riparian habitats remain vulnerable to impacts from projected population growth, development, invasive species, climate change and sea level rise (California Department of Fish and Game 2007, Point Reyes Bird Observatory Conservation Science 2011). Land use, specifically development within and adjacent to wetland and riparian areas, is a principal cause of habitat loss and degradation. According to the California Fish and Game Commission Wetlands Resources Policy, "Projects which impact wetlands are damaging to fish and wildlife resources if they result in a net loss of wetland acreage or wetland habitat value."

As described in this review, the scientific literature establishes that certain conservation strategies and mitigation methods are likely to be effective in protecting and minimizing development and land use-related impacts to wetland and riparian habitats. Individual development projects can have site-specific and cumulative effects on adjacent habitats; however, land use is the major driver of freshwater ecosystem conditions (Allan 2004, Langpap et al. 2008, Tockner et al. 2008). Land use activities and intensity profoundly affect riverine and other freshwater aquatic habitats on a watershed, regional, and global scale though habitat conversion and fragmentation, increasing road density, alterations of peak flows and floods, degradation of soil and water, increases in nutrient and pollution inputs, spreading invasive species, wildfire suppression, and altering local climate (Ziemer and Lisle 1998, Theobald et al. 2005, Allan 2004, Foley et al. 2005, Stein et al. 2005). For instance, in a detailed, site-specific analysis, California Department of Forestry and Fire Protection (2003) found that whereas only 4 percent of natural habitat in El Dorado County, CA. was lost to development, nearly 40 percent had greatly reduced habitat quality.

Habitat destruction through land use alterations is generally considered a primary cause of species endangerment (Wilcove et al.1998). Consequently, to achieve the long-term maintenance of wetland, riparian, and riverine habitats, effective watershed, regional, or landscape-level planning that addresses the ecological effects of land use is equally as important as protecting these habitats from the direct impacts of adjacent development (Michalak and Lerner 2007).

The California Fish and Game Commission's Land Use Planning Policy (California Fish and Game Commission 2013b) recognizes the importance of land use planning in the conservation of California's fish and wildlife. To provide maximum protection of fish and wildlife, this policy directs the Department to: 1) promote the development of regional conservation planning, 2) review, coordinate and provide comments and recommendations on federal, state, local planning efforts, and 3) participate in local land use planning processes for the purpose of conserving and protecting fish or wildlife habitat (California Fish and Game Commission 2013b). Other landscape level planning approaches employed by the Department to protect wetland and riparian habitats include implementation of California's Wildlife Action Plan (California Department of Fish and Game 2007) and the Recovery Strategy for California Coho Salmon (California Department of Fish and Game 2004) and participation in the Riparian Bird Conservation Plan (Riparian Habitat Joint Venture 2004).

IMPORTANCE OF WETLAND AND RIPARIAN HABITATS

California's Wetland and riparian habitats are essential for a wide variety of important resident and migratory fish and wildlife species (California Department of Fish and Game 2001, Riparian Habitat Joint Venture 2004, California Department of Fish and Game 2007). The role of riparian habitat in supporting biodiversity is well documented and its relative ecologic importance greatly exceeds the proportion of the landscape it occupies (Allan and Flecker 1993, Naiman et al.1993, 2000, Crow et al. 2000, Dahl 2000). According to Naiman et al. (1993, 2000), natural riparian corridors are the most diverse, dynamic, and complex terrestrial habitat type, and thus, they play an essential role in conserving regional biodiversity.

Because of their seasonal or year-round water supply, cool microclimate, productivity, nutrient cycling and food availability, wetlands and riparian habitats are vital to the majority of California's wildlife species (California Department of Fish and Game 2007). According to the California Fish and Game Commission Wetlands Resources Policy, "Wetland habitat is also recognized as providing habitat for over half of the listed endangered and threatened species in California." Wetlands are required by 50 percent of animals and 28 percent of plants listed pursuant to the federal Endangered Species Act (ESA) (Niering 1988). In the Pacific Coast Ecoregion, which includes much of the Department's Northern Region, 60 percent of amphibian species, 16 percent of reptiles, 34 percent of birds, and 12 percent of mammals can be classified as riparian obligates (Kelsey and West 1998, in Naiman et al. 2000). Wetlands and riparian corridors also serve as important wildlife migration and dispersal routes for both aquatic and terrestrial wildlife.

Riparian areas provide an ecological linkage or transition between aquatic and terrestrial habitats and directly affect the delivery, routing, and composition of water, nutrients, sediment, and wood into and through a stream system (Franklin 1992, Naiman et al.1993, Crow et al. 2000, Naiman et al. 2000, Bolton and Shellberg 2001). Recurrent flooding in riparian habitats results in frequent disturbances related to episodic or chronic inundation, sediment transport, and the abrasive and erosive forces of the transport of water, large wood, and bedload (Naiman and Decamps 1997, Crow et al. 2000). Seasonal or continual water availability and regular nutrient inputs also create an especially fertile and productive floodplain habitat (Naiman et al.1993, 2000, Crow et al. 2000). The combination of these processes, in turn, creates habitat complexity and variability in time as well as in space, resulting in ecologically diverse plant and animal communities (Franklin 1992, Naiman and Decamps 1997, Crow et al. 2000, Robinson et al. 2002).

California's wetland and riparian habitats are also important for the valuable ecosystem services they provide and the many recreational opportunities they offer. For instance, wetlands and floodplains store and meter floodwaters, recharge groundwater aquifers, trap sediment, filter pollution, help minimize erosion, lessen peak flow velocities, and protect against storm surges (Mitsch and Gosselink 2000, Tockner et al. 2008). In doing so, they protect adjacent upland, down-stream, and coastal properties from loss and damage during flooding and help maintain surface and groundwater during summer months. These habitats are also popular destinations for people that enjoy camping, fishing, hunting, boating, wildlife viewing and other outdoor recreational activities. According to the National Research Council (2002), "because riparian areas perform a disproportionate number of biological and physical functions on a unit area basis, their restoration can have a major influence on achieving the goals of the Clean Water Act, the Endangered Species Act, and flood damage control programs."

Birds

California's wetland and riparian habitat has been identified as the most critical habitat for conserving Neotropical migrant birds (California Department of Fish and Game 2003, Riparian Habitat Joint Venture 2004). Of the 63 bird taxa designated as California Species of Special Concern (SSC), 27 taxa (43 percent) primarily utilize wetland habitats and another 11 taxa (17 percent) are riparian forest inhabitants (Shuford and Gardali 2008). SSC are designated by the Department for species with declining population levels, limited ranges, and/or continuing threats that make them vulnerable to extinction. Though not listed pursuant to the ESA or CESA, the goal of designating taxa as SSC is to halt or reverse their decline by calling attention to their plight and addressing habitat conservation issues early enough to secure their long-term viability. A combined total of 60 percent of California's bird SSC are dependent upon wetland and riparian habitats, demonstrating both ecological importance and the threat to these habitats. The greatest factor in the decline of the willow flycatcher (*Empidonax traillii*), a State-endangered species, is the extensive loss, fragmentation, and modification of riparian breeding habitat (Bombay et al. 2003). Likewise, wetland loss is the principal threat to the State-threatened greater sandhill crane (*Grus canadensis tabida*) (Meine and Archibald 1996).

Fish

Native fish populations are dependent upon healthy aquatic ecosystems (Moyle 2002). Wetland, riparian vegetation, and associated floodplain provide many essential benefits to stream and river fish habitat (Moyle 2002, California Department of Fish and Game 2007). These features influence channel geomorphology and stream flow by providing channel roughness, bank stability, habitat heterogeneity and complexity. Riparian forests provide thermal protection, shade, and large woody debris. Large woody debris stabilizes substrate, provides shelter and cover from predators, facilitates pool establishment and maintenance, maintains spawning bed integrity, and creates habitat for aquatic invertebrate prey. Wetland and riparian areas also provide critical fish habitat in the form of off-channel and back-water winter-rearing sites and floodwater refugia (California Department of Fish and Game 2007).

Fish across North America are under severe ecological pressure from land use changes. During the past century, three genera, 27 species, and 13 subspecies of North American fish have become extinct (Miller et al. 1989). Habitat loss and introduced species were the most common factor responsible for these extinctions, 73 percent and 68 percent of cases, respectively, followed by chemical pollution (38 percent) (Miller et al.1989). According to Williams et al. (1989), approximately one out of three North American freshwater fish species and subspecies are now either threatened, endangered, or deserving of special consideration.

In California, complex and resilient natural ecosystems for fish are being replaced by simplified, highly altered systems that are unpredictable in structure and dominated by non-native species (Moyle and Williams 1990, California Department of Fish and Game 2004, 2007). Habitat loss and modification including loss of riparian forest, and increased water pollution, non-native species, and water diversions are some of the most significant factors negatively affecting California's native fishes (Moyle 2002). These impacts are the result of numerous human activities, including mining, logging, road construction on unstable slopes, over-grazing and urban/exurban development (Moyle 2002). More recently, large-scale outdoor marijuana cultivation in northern California has also been documented as having substantial negative impacts on fish and other aquatic and terrestrial wildlife species (Department unpublished data).

The threats to California's fishes mirror those to other North American taxa. Of California's 67 native inland fish species, seven (10 percent) are extinct in the state or globally; 13 (19 percent) are State or federally listed (as of 2001), and 19 (29 percent) are listed as SSC (Moyle 2002). Few fishes have been more significantly impacted by loss and alteration of habitat than Pacific salmon and anadromous trout (Moyle 2002). These species, including Chinook salmon (Oncorhynchus tshawytscha), coho salmon (O. kisutch), steelhead trout (O. mykiss), and coastal cutthroat trout (O. clarkii) are vitally important ecological and economic keystone species in California. Coho salmon, for instance, has undergone at least a 70-percent decline in abundance since the 1960s, and is currently at 6 to 15 percent of its abundance during the 1940s (California Department of Fish and Game 2004). If present population trends continue, Katz et al. (2012) anticipate 25 (78 percent) of California's 32 native salmonid taxa will likely be extinct or extirpated within the next century.

Amphibians and Reptiles

California's aquatic habitats and their adjacent uplands are essential habitat for numerous aquatic and semi-aquatic amphibian and reptile species. The Department's

Special Animal List recognizes 45 amphibian species as listed pursuant to CESA, ESA, or some other "watch list" including SSC. Of these 45 species, 13 occur in the Department's Northern Region. Ten of these 13 species rely upon streams and wetlands for breeding habitat and adjacent upland habitat for other critical life functions.

Some of Northern Region's numerous amphibian and reptile SSC, which rely upon wetland and riparian habitats include: southern torrent salamander (*Rhyacotriton variegatus*), Del Norte salamander (*Plethodon elongatus elongatus*), Cascades frog (*Rana cascadae*), northern red-legged frog (*R. aurora*), California red-legged frog (*R. draytonii*), Oregon spotted frog (*R. pretiosa*), foothill yellow-legged frog (*R. boylii*), coastal tailed frog (*Ascaphus truei*), and Pacific pond turtle (*Actinemys marmorata*).

Amphibians are currently undergoing a global collapse (Lannoo 2005, Wake and Vrendenburg 2008). On a regional and state-wide scale numerous amphibian species and populations are also documented in decline (Fellers et al. 2008). For instance, because of the decline in aquatic habitat types, half of 29 native amphibian species in the Sierra Nevada and Cascades Region are at risk of extinction (California Department of Fish and Game 2007). While this decline appears to have multiple causes, habitat loss and fragmentation are now considered among the greatest threats to amphibian populations (Lannoo 2005, Cushman 2006).

All 47 amphibian species occurring in the Pacific Northwest are either facultative or obligate stream-riparian associates (Olson et al. 2007). Ninety percent of these occur in forested habitats and about a third are stream-riparian obligate species (Olson et al. 2007). Of particular conservation significance is that a quarter of these forest-dwelling amphibians are tied to smaller headwater streams (Olson et al. 2007). Despite substantial evidence that small headwater streams are important to amphibian populations, as well as providing vital ecosystem services to downstream watersheds, small headwater streams face the most substantial threat of elimination by urbanization (Elmore and Kaushal 2008). Also, compared to larger stream types, small headwater streams typically receive the narrowest streamside buffers in local and state-wide mitigation approaches, e.g. the California Forest Practice Rules (California Department of Forestry and Fire Protection 2012).

<u>Bats</u>

The loss and fragmentation of quality foraging habitat is a major threat to bat populations worldwide (Racey and Entwistle 2008). Populations of many bat species in North America and globally are declining and currently approximately 25 percent of the global bat fauna are listed as threatened by the International Union for Conservation of Nature. According to the California Natural Diversity Database (CNDDB), 12 of California's 25 bat species are designated SSC, USDA Forest Service Sensitive, or federally Endangered.

Many North American bat species forage near or directly over open water, while others feed in a variety of habitats but are often associated with riparian vegetation (Pierson 1998). All 15 bat species occurring on northern California and the Pacific Northwest are insectivorous. Riparian habitats are of disproportionate importance for many bat species because they are insect-rich environments and provide roosting, foraging sites, and drinking water (Wunder and Carey 1996, Grindal et al. 1999). As such, bats were identified as an important species group whose conservation justified enhanced riparian buffer protection in the management guidelines of the federal Northwest Forest Plan (Seidman and Zabel 2001).

In Douglas-fir forests of the Washington Cascade Range and the Oregon Coast Range, foraging activity of *Myotis* bat species was found to be 10 times greater over water than within the forest interior (Thomas 1988). In coastal British Columbia, the little brown bat (*Myotis lucifugus*) was found to be 75 times more active over lakes and ponds than in forested habitat (Lunde and Harestad 1986). In northern California, Seidman and Zabel (2001) found substantial foraging utilization on intermittent streams, and importantly, streams with discontinuous flows had similar levels of bat activity as streams with continuous flowing or standing water.

Drastic population declines of the formerly abundant cave myotis (*Myotis velifer*) from California's Colorado River basin (Pierson 1998) and the western red bat (*Lasiurus blossevillii*) from the Sacramento and San Joaquin River basins are associated with the loss of cottonwood-dominated riparian forests. According to Pierson et al. (2006), the western red bat in California would greatly benefit from riparian restoration, particularly the recruitment of cottonwood/sycamore forests and the reinstatement of natural flood regimes. According to Ober and Hayes (2008), the best strategy for Pacific Northwest bat fauna conservation over broad spatial scales is the maintenance or creation of a diversity of riparian vegetation conditions.

<u>Sensitive Plants and Natural Communities</u>
Northern California is globally renowned as a biodiversity hotspot for rare, endemic, and unusual plants, many of which are associated with aquatic habitats. Because of this botanical diversity, the Klamath-Siskiyou Ecoregion, which encompasses much of northwestern California, has been named an Area of Global Botanical Significance (one of seven in North America) by the World Conservation Union (Ricketts et al. 1999). It has also been proposed as a World Heritage Site and UNESCO Biosphere Reserve (Ricketts et al. 1999). Sacramento River Valley vernal pools, for example, are globally unique habitats and threatened throughout their range while the Northern Region's coastal lagoons and peatlands are some of the largest in the state and recognized botanical hotspots (Leppig 2004).

Northern California's diverse wetland and riparian habitats are home to more than 116 sensitive plant species. According to the CNDDB, approximately one third of the region's sensitive plant species occur in aquatic and riparian habitats. In northern California's redwood forests, 40 percent of sensitive plant species occur in wetland and riparian habitats (Golec et al. 2006). According to Comer et al. (2005), California has more at-risk plant species occurring within isolated wetlands (143 species) and also more plant species listed pursuant to the ESA (32 species) tied to isolated wetlands, than any other state.

Certain vegetation types or natural communities are rare or threatened in their own right and thus have ecological importance and conservation status in addition to the environmental services and wildlife habitat they provide. The CNDDB classifies vegetation for the primary purpose of assisting in determining the state-wide significance and rarity of various vegetation types. CNDDB first classifies vegetation types into specific "alliances" based upon species dominance. CNDDB ranks these alliances based upon their rarity and threat. Alliances designated with a State (S) ranking of S1, S2, and S3 are considered rare and of high priority for inventory. See http://www.dfg.ca.gov/biogeodata/vegcamp/natural_comm_list.asp for more detailed information on how CNDDB addresses natural communities.

In northern California, 16 riparian vegetation alliances, dominated by alder (*Alnus viridis*), willow (*Salix* spp.), birch (*Betula* spp.), and cottonwood (*Populus* spp.), have State rankings of S2 or S3 (CNDDB 2013). Numerous other non-woody wetland vegetation types in the region also have State rankings of S1 to S3. The high number of riparian vegetation alliances designated as rare and of high priority for inventory is another indication of the ecological significance of these habitats and the need for effective conservation strategies to prevent their further loss and degradation.

DEVELOPMENT AND LAND USE IMPACTS

Development can result in permanent wetland and riparian habitat loss through conversion to non-habitat, and conversion of wetlands to uplands. In addition to direct habitat loss, development also has three principal indirect effects on adjacent habitat: 1) fragmentation of habitat into smaller, non-contiguous areas of less-functional habitat by structures, roads, driveways, yards and associated facilities; 2) the introduction or increased prevalence of exotic species or species that are habitat generalists, termed "human adapted" or "urban exploiters," and 3) decreases in native species abundance and biodiversity and the loss of "human-sensitive" species that require natural habitats (Davies et al. 2001, Hansen et al. 2005, California Department of Fish and Game 2007). In general, these effects occur because development tends to favor species well-adapted to human habitation with subsequent negative effects on sensitive species and those species best adapted to natural habitats (Marzluff and Neatherlin 2006). For example, numerous studies document how human activities in natural areas disturb bird populations and reduce bird diversity, abundance, and reproductive success (Rodgers et al. 1997, Fernandez-Juricic 2002, Burger et al. 2004, Banks and Bryant 2007).

Even low-density residential development can result in habitat loss and degradation because: 1) structures require 100-foot-wide defensible space fire-safe buffers—which necessitates vegetation clearing around them (Pub. Resources Code § 4291), 2) local wildlife populations' response to development can continue several decades after habitat alteration or construction (Hansen et al. 2005), and 3) in addition to local effects, development has been shown to alter the ecological processes and biodiversity in areas more far-removed from the development, including in parks, preserves, and national forests (Hansen et al. 2005, Johnston and Klemens 2005b). Other studies have also demonstrated that the surrounding landscape “matrix” and the amount of urbanization can strongly influence riparian and wetland species even if development is not directly adjacent to these habitats (Rodewald and Bakermans 2006, Roe and Georges 2007).

Additional adverse effects from development adjacent to wetland habitats include: vegetation removal; water diversions and altered hydrology; diminished water quality from the discharge of pollutants such as sediment, toxic substances, and pathogens; disturbance to wildlife from pets, noise, and human activities; filling and refuse dumping; and altered microclimate. Human development also negatively impacts wildlife through increased road-kill (Trombulak and Frissell 2000, Malo et al. 2004, Beebee 2013), light pollution (Longcore and Rich 2004, Rich and Longcore 2006), the killing of and disturbance to wildlife by domestic and feral animals such as house cats, and increased human conflict with wildlife such as black bear, mountain lion, and fox. Development in close proximity to natural areas often provides attractive nuisances such as orchards, gardens, pets, compost bins, and garbage receptacles, which results in human-wildlife conflicts that often resulting in the killing (depredation) of these animals. For an in-depth review of the impacts of land use and urbanization on stream ecosystems see Paul and Meyer (2001) and Allan (2004).

Development-related loss of native species abundance and diversity or the increase in exotic and native generalist species has been shown for bird assemblages (Beissinger and Osborne 1982, Wilcove 1985, Luginbuhl et al. 2001, Odell et al. 2003), mammals (Maestas et al. 2001), fish (Paul and Meyer 2001), amphibians (Davidson et al. 2001, Ridley et al. 2005), terrestrial and freshwater invertebrates (Miyashita et al. 1998, Paul and Meyer 2001), and plants (Galatowitsch et al. 1999, Mack and Lonsdale 2001, Reichard and White 2001).

Many studies have shown that habitat fragmentation from urban and exurban development and other human activities results in significant declines in species richness in a broad range of avian communities (Wilcove 1985, Engels and Sexton 1994, Marzluff 2001, Hansen et al. 2005). Rottenborn (1999) found that urbanization on lands adjacent to intact riparian woodlands has substantial impacts on riparian bird communities. Human-adapted corvids (ravens, crows, and jays) are effective nest predators whose abundance has increased dramatically due to urbanization in western North America and worldwide in the last century (Luginbuhl et al. 2001). Increased nest predation by corvids and other human-adapted species has had a significant effect on bird populations adjacent to urbanized areas (Wilcove 1985, Engels and Sexton 1994, Marzluff 2001, Odell et al. 2003, Hansen et al. 2005).

Bank erosion is a fundamental riverine process that drives lateral channel migration, thus creating and maintaining off-channel habitats, affecting recruitment of sediment and large woody debris and acting as a key regulator of aquatic habitat in the main stem and riparian habitat in the floodplain (USFWS 2004, CALFED 2008). Development often leads to streambank stabilization, which prevents bank erosion and channel migration. Bank stabilization (also known as armoring) such as placement of revetment (large boulders known as rip-rap) and efforts to dredge channels or build flood-control levees, commonly occur where development is placed in flood plains or too close to stream and river channels and is then threatened by flooding and bank erosion.

Revetment can negatively impact riparian vegetation, stream bank morphology, stream flow characteristics and aquatic and terrestrial habitat quality (USFWS 2000, USFWS 2004). Revetment can eliminate structural bank features such as large wood and overhanging banks and vegetation, which provide fish with refuge from high flows, needed habitat complexity, and cover from potential predators (Peters et al. 1998, USFWS 2000, USFWS 2004). In redwood forests, stream reaches with revetment were shown to have lower plant species richness, vegetation cover, and tree seedling density compared with streambanks without revetment (Russell and Terada 2009).

Loss of California's wetland and riparian habitats has resulted in many water quality impairments. For example, removal of riparian vegetation is a contributing factor to impairment of over three quarters of the water bodies listed by the North Coast Regional Water Quality Control Board pursuant to the Clean Water Act Section 303(d)(SWRCB 2002). According to SWRCB (2002), more than 50 water bodies in northern California, including reaches of almost all major river systems, are listed pursuant to Clean Water Act section 303(d) as impaired for any of the following reasons: temperature, sedimentation/siltation, nutrients, and bacteria.

Impacts to Amphibians and Reptiles

Amphibians appear to be particularly vulnerable to habitat loss and fragmentation due to multiple factors (Carr and Fahrig 2001, Houlahan and Findlay 2003, Cushman 2006). These factors include: 1) relatively short distances traveled; 2) high vulnerability to death when moving across roads and through inhospitable terrain; 3) narrow habitat tolerances; and 4) high vulnerability to pathogens, invasive species, climate change, increased ultraviolet-B exposure and environmental pollution (Gibbs and Shriver 2005, Cushman 2006, see also Olson et al. 2007).

Traffic-caused mortality ("road-kill") is a major cause of amphibian mortality and may contribute to their global decline (Fahrig et al. 2005, Gibbs and Shriver 2005, Glista et al. 2008). Amphibians may be especially vulnerable to traffic mortality because they often migrate *en masse* to and from breeding wetlands (Glista et al. 2008) and because of their relatively slow speed. For example, as reported in Fahrig et al. (2005) and Ehmann and Cogger (1985) a conservative estimate of 5,480,000 reptiles and amphibians are killed annually by traffic in Australia. Rosen and Lowe (1994) estimate that tens or hundreds of millions of snakes have been killed by automobiles in the United States since the advent of the automotive age.

Development adjacent to wetlands and riparian areas can eliminate native habitat for amphibians and reptiles. Until recently, wetlands and streams were thought to be core habitat for semi-aquatic amphibians and reptiles, and adjacent riparian and upland habitats were considered merely a buffer zone to protect aquatic habitat (Semlitsch and Jensen 2001). However, many reptiles and amphibians that utilize wetlands and streams for reproduction and juvenile life stages depend upon, and range widely into, adjacent uplands as adults (Semlitsch and Bodie 2003, Rittenhouse and Semlitsch 2007, Harper et al. 2008). Numerous studies have documented the critical importance of upland areas for migration and adult habitat for amphibians and reptiles that breed in insolated wetlands (Joyal et al. 2001, Gibbons 2003, Semlitsch and Bodie 2003, Cushman 2006, Denoel & Ficetola 2008). In an analysis of six North American salamander species, Semlitsch (1998) determined that 125 meters (410 feet) was the mean distance individuals were found from the edge of aquatic habitats and that state and federal wetland protections do not take into account the wide upland use of these aquatic organisms.

The western pond turtle is a case in point. It is listed as state endangered in Washington, Sensitive–Critical in Oregon, and a SSC in California. Primary threats to this species are loss and alteration of both aquatic and terrestrial habitats (Bury and Germano 2008). However, in no state is its upland habitat effectively protected (Bury and Germano 2008). Uplands adjacent to aquatic habitats are critical to this species, as some individuals can spend as much as seven months of each year on land as reported by Reese and Welsh (1998) in Bury and Germano (2008). According to this study, females lay eggs as many as 400 meters (1,312 feet) from streams, however most nests are within 50 meters (164 feet) of the water's edge.

In northern California, many amphibians, including the federally threatened California red-legged frog, the Cascades frog, northern red-legged frog, and foothill yellow-legged frog, all depend upon upland habitats for adult life stages (Bulger et al. 2003). In a California red-legged frog study, Bulger et al. (2003) found adults as far as 500 meters (1,640 feet) from water. These researchers suggest adequate protection around California red-legged frog breeding sites can be achieved by maintaining at least 100 meters (328 feet) of suitable habitat around wetlands. To prevent the imminent regional extirpation of the Cascades frog, Fellers et al. (2008) recommend restricting habitat alterations within proximity to their breeding grounds. In a study of Appalachian salamanders, Crawford and Semlitsch (2007) found that 95 percent of salamanders occupied a core terrestrial habitat within 27 meters (89 feet) of a stream.

Harper et al. (2008) determined that regulations protecting 30 meters (98 feet) or less of surrounding terrestrial habitat are inadequate to support viable populations of pool-breeding amphibians. A summary of data on use of terrestrial habitats by wetland-associated amphibians and reptiles found that core habitats ranged from 159 to 290 meters (521 to 951 feet) for amphibians and 127 to 289 meters (416 to 948 feet) for reptiles from the edge of aquatic habitats (Semlitsch and Bodie 2003). A California tiger salamander (*Ambystoma californiense*) habitat analysis determined 50 percent of an adult population was found greater than 150 meters (492 feet) from a breeding pond (Trenham and Shaffer 2005).

Thus riparian or upland habitat surrounding wetlands and streams is documented to function as essential and core habitat for many aquatic and riparian-dependent amphibian and reptile species and should not be viewed merely as a disturbance buffer for aquatic habitat from surrounding land-use practices (Semlitsch and Jensen 2001).

The conservation of reptile and amphibian biodiversity is one of the most compelling biological reasons to protect small, isolated wetlands (Batzer et al. 2006) and to implement wider and more effective upland buffers adjacent to aquatic habitats.

Fragmentation and Altered Microclimate

Riparian habitat adjacent to streams and rivers has a controlling influence on microclimate characteristics of the stream corridor (FEMAT 1993). Land management, such as removal of forest canopy creates an "edge effect" that results in microclimate changes to the remaining habitat, including changes in relative humidity, solar radiation, soil and water temperature, average high and low ambient temperatures, and wind velocity within forested areas adjacent to forest openings (FEMAT 1993, Davies et al. 2001). Riparian microclimate, which is often more cool and moist than adjacent habitats, together with stream temperatures, are important and related habitat characteristics of stream and river ecosystems (Franklin 1992, Moore et al. 1995).

Numerous studies on the edge effects and fragmentation from adjacent land use, (especially forest removal) have documented significant indirect biotic and abiotic impacts on remnant habitats (FEMAT 1993, Brosofske et al. 1997, Chen et al. 1999). For example, microclimate changes in remnant habitat patches resulting from adjacent land use practices have been documented to extend from 15 meters (50 feet) to greater than 250 meters (820 feet) into remnant patches (Jules and Rathcke 1999, Gehlhausen et al. 2000, Zheng 2000, Davies et al. 2004, Concilio 2005).

Impacts to vegetation structure in remnant forest patches adjacent to forest openings have been documented to include changes in species density, growth rate, volume, above- and below-ground biomass, and vegetation height (Brothers 1993, Fraver 1994, Malcolm 1994, Young and Mitchell 1994, Lovejoy et al. 1996, Laurance et al. 1998, Stinton et al. 2000, Franklin et al. 2004, Harper et al. 2005).

Edge effect changes in vegetation composition in adjoining remnant forests, including species composition, species richness, and plant community have been documented by Russell and Jones (2002), Benito-Malvido and Martinez-Ramos (2003), Moen and Jonsson (2003), Watkins et al. (2003), Harper et al. (2005), Halpern et al. (2005), Nelson et al. (2005a), and Nelson et al. (2005b). While changes to plant life history and plant/animal interactions in forest fragments, including survival, growth, development, reproduction, pollination, seed set and dispersal are documented by Jules and Rathcke (1999), Ozanne et al. (2000), Tallmon et al. (2003), and Nelson and Halpern (2005a).

In their study sites in coniferous forests in Washington State, Brosofske et al. (1997) concluded that a minimum 45-meter buffer width (147 feet) on both sides of a stream (90 meter (295 feet) total buffer width) is necessary to maintain a natural riparian microclimate along streams. Changes in humidity and wind speed were documented to extend greater than 240 meters (787 feet) from clear-cut edges into an old-growth Douglas-fir forest (Chen et al. 1995). Based upon the studies cited here and elsewhere, there is strong evidence that adjacent land use can have a significant indirect effect on the microclimate and vegetation characteristics of wetland and riparian habitats, especially if the land use entails habitat conversion or removal of forest vegetation.

Domestic and Feral Cats

The scientific literature suggests wildlife would substantially benefit if proposed residential development adjacent to wetland and riparian habitats required that domestic housecats be kept indoors. Winter and Wallace (2006) report there are at least 90 million pet cats in the United States and perhaps an equal number of feral cats. Free-roaming cats in the United States annually kill millions of birds, mammals, amphibians and reptiles—including endangered species—and predation by feral and free-ranging house cats is now considered one of the greatest threats to avian biodiversity (Winter and Wallace 2006). Domestic cats are considered primarily responsible for the extinction of 33 bird species world-wide since the 1600s (Winter and Wallace 2006).

During a five month period, Great Britain's estimated nine million domestic cats brought home (killed) an estimated 92 million prey items, including 57 million mammals, 27 million birds, and five million reptiles and amphibians (Woods et al. 2003). According to the Florida Fish and Wildlife Conservation Commission (2003), Florida has an estimated 5.3 million owned and un-owned (feral) domestic cats that are sometimes outdoors. It is estimated Florida cats annually kill millions of wildlife prey and that their ecological impact is best documented and has the most damaging effect on endangered species and other taxa with small population sizes or limited distributions (Florida Fish and Wildlife Conservation Commission 2003).

In California, cat predation is a threat to numerous sensitive bird species, including California least tern, western snowy plover, California black rail, burrowing owl, and tricolored blackbird (Winter and Wallace 2006). A study of the impacts of residential development adjacent to fragmented natural habitats in San Diego County showed: 32 percent of residences owned cats; each residence had on average 1.7 cats; 78 percent let their cats outdoors; and 84 percent of outdoor cats brought back kills to the residence (Crooks and Soule 1999). In this study, cat owners with outdoor cats that hunted returned on average 24 rodents, 15 birds, and 17 lizards to the residence annually. In addition to direct predation of birds, domestic cats have been shown to have substantial sub-lethal effects on birds through the loss of reproductive capacity (reduced fecundity) (Bonnington et al. 2012).

Night Light Pollution

Artificial light is a consequence of development. Roads and buildings typically include exterior night lighting and have the potential to introduce light pollution to adjacent wetland, marine, and riparian habitats. Adverse ecological effects of artificial night lighting on terrestrial, aquatic, and marine resources such as fish, birds, mammals, and plants are well documented (Johnson and Klemens 2005a, Rich and Longcore 2006). Some of these effects include altered migration patterns and reproductive and development rates, changes in singing behavior in bird species Miller (2006), changes in foraging behavior and predator-prey interactions, altered natural community assemblages, and phototaxis (attraction and movement towards light), disorientation, entrapment, and temporary blindness (Longcore and Rich 2004). The Department has determined that artificial night lighting can significantly affect marine and near-shore wildlife (California Department of Fish and Game 2007). Light pollution disrupts the abilities of night-foraging birds, renders seabirds more vulnerable to predation, and has resulted in nest abandonment and low reproductive success for brown pelicans (California Department of Fish and Game 2007). Johnston et al. (2004) list artificial lighting as a permanent impact to bat roosts and recommend that artificial lighting be directed away from bat roosts or possibly shaded by trees.

In an experimental study, Becker et al. (2013) found that artificial light associated with human-made structures has the potential to alter fish communities within urban estuarine ecosystems by creating optimal conditions for predators. Future coastal development should consider the ecological implications of lighting on aquatic communities. They recommend lighting be minimized around coastal infrastructure and the use of red lights, which have limited penetration through water, be considered (Becker et al. 2013). Research on the effects of artificial lighting on salmonid populations indicate that increased light intensity appears to slow or stop out-migrating juvenile salmon and affects feeding patterns. Juvenile salmonids in the presence of increased artificial night lighting may be more vulnerable to predation (McDonald 1960, Patten 1971, Ginetz and Larkin 1976, Tabor et al. 2004).

Stormwater Runoff Pollution

Non-point source pollution found in urban stormwater runoff is recognized as a leading threat to the nation's water quality (USEPA 1999). Two comprehensive assessments of the nation's management of oceans and coastal resources determined non-point source pollution is one of the most significant emerging threats to aquatic species and that non-point source pollution represents the greatest pollution threat to oceans and coasts (Pew Ocean Commission 2003, U.S. Commission on Ocean Policy 2004). According to the California Department of Fish and Game (2007), 40,000 tons of contaminants (heavy metals, pesticides, fertilizers, polychlorinated biphenyls, etc.) enter the Bay Delta annually from urban and agricultural runoff.

Urbanization and other forms of development increase the runoff of pollutants from terrestrial landscapes to aquatic habitats. Non-point source pollutants in stormwater runoff, such as petroleum products, metals, pathogens, nutrients, pesticides, and domestic animal feces are well documented as having acute and chronic lethal and sub-lethal effects on salmonids and other aquatic organisms. For instance, polycyclic aromatic hydrocarbons, a byproduct of petroleum use, are a pervasive component of stormwater runoff and are documented to have numerous detrimental health effects on salmonid fishes (Incardona et al. 2004, 2005). These contaminants originate from commercial, industrial, residential and agricultural land uses and are mobilized from roads, roofs, farms, lawns, crops and other surfaces and transported by rainwater to aquatic habitats. Stream habitat quality has been shown to degrade when impervious surfaces, such as buildings, roads, and parking lots cover greater than 10 percent of a watershed, with severe degradation expected beyond 25 percent (Arnold and Gibbons 1996, Watershed Protection Research 2003). In West Virginia, biological integrity ratings of poor or very poor occurred in catchments with less than 7 percent urban land use (Snyder et al. 2003).

Moore and Palmer (2005) studied invertebrate biodiversity in 29 small headwater streams in rapidly urbanizing agricultural lands near Washington, D.C. They made two significant findings with important conservation implications: 1) Invertebrate biodiversity was extremely high in agricultural headwater streams and progressively declined along a land-use gradient toward urbanization. 2) In urban streams, there was a strong positive relationship between intact riparian forest buffers and in-stream biodiversity. This suggests agricultural preservation programs (e.g. regional conservation approaches that include sufficient buffers and other appropriate conservation measures) are likely important to conserve freshwater biodiversity in urbanizing areas. Their study suggests that efforts to preserve and restore urban riparian buffers may mitigate some of the impacts of urbanization on watersheds, even where there is a substantial amount of development. They conclude that from a biodiversity perspective, headwater streams in areas already highly urbanized should not be viewed as “lost causes” given the on-site and down-stream ecosystem services they provide (Moore and Palmer 2005).

Essentially any surface which does not have the capability to pond and infiltrate water will produce runoff during storm events. When a land area is altered from a field, farm or forest ecosystem to an impervious urbanized land use the hydrology of the system is altered. Water, which was previously infiltrated into the soil and converted to groundwater, utilized by plants and evaporated or transpired into the atmosphere, is now converted directly into surface runoff (USEPA 2000, OPR 2009).

As watersheds urbanize, streams receive larger volumes of stormwater runoff, which results in a greater frequency and intensity of flood events (USEPA 2000, Office of Planning and Research 2009). This in turn often leads to stream channel instability, channel widening and scour, and the introduction of larger amounts of sediment to urban streams. Visible impacts include eroded and exposed stream banks, fallen trees, sedimentation, and recognizably turbid conditions. The increased flooding frequency of urban areas also poses a threat to public safety and property. In the Russian River watershed of Sonoma County, Lohse et al. (2008) found that even low-density exurban development resulted in fine sediment inputs to streams, which negatively and significantly degraded endangered salmon spawning and rearing habitat. Both water quality and water quantity impacts associated with stormwater runoff combine to degrade stream habitats.

Paved surfaces such as roads, parking lots, and driveways create effective conduits for oil, grease, and other toxic pollutants to enter into coastal waters. Every eight months, nearly 11 million gallons of oil runs off streets and driveways into the nation’s waterways (Pew Ocean Commission 2003). Rooftops, another fixture of development and urbanization, are also known to be both sources and pathways for contaminated runoff (Van Metre and Mahler 2003). Metal roofing, for instance, is a source of cadmium and zinc, while asphalt shingles are a source of lead. Asphalt shingle and galvanized metal roofs can leach numerous contaminants to stormwater, but they also catch and deliver fallout from airborne contaminants released from vehicles, such as copper from brake pads and cadmium from tires (Van Metre and Mahler 2003). Concentrations of zinc and lead from rooftop runoff samples have been shown to exceed established sediment quality guidelines for probable toxicity of bed sediments to benthic biota (Van Metre and Mahler 2003).

Motor vehicles are a major source of toxic stormwater contaminants such as copper, a metal that originates from vehicle exhaust and brake pad wear and is then transported to aquatic habitats via stormwater runoff. Dissolved copper has become a common non-point source pollutant in urbanized watersheds and is now a widely distributed contaminant in lakes, rivers, and coastal marine environments (Linbo et al. 2006, McIntyre et al. 2008). Dissolved copper is neurotoxic to fish and is especially known to interfere with the normal function of the peripheral olfactory nervous system (McIntyre et al. 2008). Copper-containing stormwater runoff from urban landscapes has the potential to cause chemosensory deprivation and increased predation mortality in exposed salmon (Sandahl et al. 2007).

California leads the nation in agricultural pesticide use and pesticides are a common component of stormwater pollution in the State. According to the California Department of Pesticide Regulation, more than 173 million pounds of pesticides were used in California in 2010 (California Department of Pesticide Regulation 2012). While agriculture is the major category of pesticide use in California subject to reporting, other commercial and residential uses such as landscaping utilize millions of pounds annually (California Department of Pesticide Regulation 2012). Pesticides are frequently detected in northern California salmon habitats (Guo et al. 2004, Scholz et al. 2006, Gilliom 2007). Transport of pesticides by surface runoff during rainfall events is a major process contributing to pesticide contamination in the Sacramento River (Guo et al. 2004). More than half of the water samples from certain drainages in California’s Central Valley contained more than seven different pesticides (Scholz et al. 2006).

Mixtures of pesticides that have been commonly reported in salmon habitats may pose a greater challenge for species recovery than previously anticipated (Domagalski et al. 2000, Laetz et al. 2009). Certain combinations of pesticides occurring in salmon streams can have additive cumulative neurotoxicity and behavioral effects on salmon under natural exposure conditions, and therefore the ecological risk of pesticide impacts on salmon recovery may be underestimated (Scholz et al. 2006). Another study concluded that several combinations of organophosphates were lethal to Pacific salmon at concentrations that were sub-lethal in single-chemical trials (Laetz et al. 2009). Exposures to low, environmentally realistic concentrations of one type of pesticide (chlorpyrifos) are closely correlated to reductions in swimming speed and feeding rates of salmon (Sandahl et al. 2005). Reductions in salmon feeding rates are likely to lead to reductions in the size of exposed salmon at the time of their seaward migration—an important determinant of individual salmon survival at sea. According to McCarthy et al. (2008), toxic stormwater runoff, urbanizing coastal streams, and coho salmon die-offs “…foreshadow potential future threats to wild salmon populations in developing watersheds in northern California and the Pacific Northwest.”

There is also ample research that links pesticides to amphibian declines (Sparling et al. 2001, Davidson 2004). Extensive experimental data show that insecticides and herbicides have profound impacts to the biodiversity and productivity of aquatic communities, including severe tadpole mortality (Relyea 2005). Organophosphate pesticides are “ubiquitous in the environment and are highly toxic to amphibians” and are directly related to their declines in California (Sparling and Fellers 2007, 2009). Their data suggest that because of pesticide use, agricultural runoff in California’s Central Valley is toxic to amphibians.

The scientific literature suggests that unmitigated stormwater runoff has a variety of negative impacts to wetland and riparian resources, and measures to reduce stormwater runoff from developed areas are imperative. Low-impact development elements such as pervious surface technologies for driveways and walkways, vegetated (green) roofs (Voelz 2006), disconnected downspouts, water gardens and vegetated swales may be used to maximize pervious surfaces and capture and maintain on-site stormwater percolation and treatment and maintain and improve the water quality of aquatic habitats (USEPA 2000). In essence, the purpose of low-impact development is to slow, spread, and sink (infiltrate into the ground) stormwater on-site to the maximum extent practicable, rather than to store, concentrate and drain stormwater off-site as quickly as possible.

By using low-impact development, projects generally seek to maintain to the greatest extent practicable, post-project pervious surfaces and minimize off-site stormwater runoff (USEPA 2000, OPR 2009. Low-impact development design elements benefit aquatic resources by: 1) filtering out pollution and increasing the quality of stormwater runoff; 2) decreasing peak flows, flood risks and erosion in downstream waters; and 3) increasing ground water recharge and therefore helping maintain biologically-important summer low flows in adjacent streams and wetlands (USEPA 2000, OPR 2009). Low-impact development has been shown to be economical, has documented effectiveness in reducing stormwater pollution, protecting stream integrity and ocean health, and therefore its use has become highly promoted (California Ocean Protection Council 2008, SWRCB 2008).

CLIMATE CHANGE IMPACTS

According to the California Global Warming Solutions Act of 2006, climate change is now considered one of the greatest threats to California’s ecosystems. Based upon current projections, by the end of this century California’s climate will be considerably warmer than today’s, snowpack will be substantially diminished, what snowpack occurs will melt much earlier in the year, and relative sea levels will have risen (California Energy Commission 2005, 2009a, 2009b, 2009c, USGCRP 2009).

California is especially vulnerable to the impacts of climate change and sea level rise because of its geographic location, long coastline, Mediterranean climate, extensive mountain and river systems, large population, and massive agricultural output (Snyder et al. 2002, Snyder and Sloan 2005, California Energy Commission 2009a, 2009c). California is also more vulnerable to climate fluctuations, relative to the rest of the U.S., because it derives a disproportionate percentage of its water supply from only a small number of winter storms, typically in the form or “atmospheric rivers” (Dettinger 2011, Dettinger et al. 2011).

In the coming decades, climate change is anticipated to exacerbate further the loss of California’s ecosystems and the services they provide (California Energy Commission 2009d). Already there is sufficient experimental and empirical evidence to generate high confidence that climate change is presently impacting wildlife species and natural systems across the globe (Parmesan and Yohe 2003, Parmesan 2006, California Energy Commission 2009d).

Regional climate models, based upon future greenhouse gas (GHG) emission scenarios, suggest warmer average temperatures and changes in precipitation patterns will occur, the total amount of water availability in California will decrease over this century, water needs will increase, and the timing of water availability will be greatly perturbed (Leung and Ghan 1999, Snyder et al. 2002, Leung et al. 2004, Snyder et al. 2004, Snyder and Sloan 2005, California Energy Commission 2009c).

These climatic changes are anticipated to result in region-wide deficits in spring and summer runoffs, which will intensify human competition for a diminishing water supply and leave natural systems and aquatic and riparian species severely impacted (Snyder et al. 2004, Snyder and Sloan 2005, Schlenker et al. 2007).

Air Temperature Increase /Extreme Heat Days

When averaged across the state, and across all GHG emission scenarios, both minimum and maximum air temperatures are projected to increase over the 21st Century (California Energy Commission 2009d). Three recent air temperature climate change projections are included in the table below.

Projected Changes in Air Temperature for Three Regions and Time Periods under Existing GHG Emission Conditions.

Region	Next Two Decades	Mid-21st Century	End of 21st Century
Klamath Basin	---	1.1 to 2.0 °C (+2.1 to 3.6 °F)	2.5 to 4.6 °C (+4.6 to 7.2 °F)
California	0.6 to 1.3 °C (~1.1 to 2.3°F)	0.8 to 2.3 °C (~1.4 to 4.1°F)	1.5 to 4.2 °C (~2.7 to 7.5°F)
Pacific Northwest	1.6 °C (+3.0 °F)	2.0 to 2.8 °C (+3.6 to 5.0 °F)	2.8 to 4.6 °C (+5.1 to 8.3 °F)

Source: California Energy Commission 2009d, Koopman et al. 2009, USGCRP 2009; Barr et al. 2010.

Water Temperature Increase

Warmer air temperatures and other effects of climate change are expected to result in higher water temperatures in California’s streams and rivers, which in turn could significantly decrease suitable habitat for some freshwater fishes (Poff et al. 2002, Mohseni et al. 2003, Yates et al. 2008, Wenger et al. 2011). Increased water temperatures reduce growth rates in fish and increase their susceptibility to disease, while warmer water also holds less dissolved oxygen, which can reduce survival in juvenile salmonids (Moyle 2002, California Department of Fish and Game 2007). Klamath River water temperature is projected to increase by approximately 2.8 to 3.3 °C (5 to 6 °F) during the 21st century (Reclamation 2011). Wenger et al. (2011) project substantial declines in habitat for trout species in the interior western United States due to climate change-related altered flow regimes and increased water temperatures. Due to a warming climate, by 2090, 25 to 41 percent of currently suitable California streams may be too warm to support trout (O'Neal 2002). In the upper Sacramento River Basin, increased water temperatures could exceed the physiological tolerances of eggs and juveniles of winter and spring run Chinook salmon (Yates et al. 2008).

<u>Decreased Snowpack, Earlier Snowmelt, Lower Spring and Summer Flows</u>
Numerous studies indicate that a warmer future climate in California will result in more winter precipitation falling as rain instead of snow and will cause snowmelt runoff to shift earlier in the season (Kim 2001, Kim et al 2002, Knowles and Cayan 2002, Snyder et al. 2002, Miller et al. 2003, Hayhoe et al. 2004, Leung et al. 2004, Snyder et al. 2004, Vanrheenen et al. 2004, Snyder and Sloan 2005).

By 2090, a projected 2.1 °C (~3.8°F) temperature increase is expected to reduce the April snowpack of the Sacramento/San Joaquin Basin by approximately half, with losses being most severe in the northern Sierra Nevada and Cascade Mountains—which would lose 66 percent of their April snowpack (Knowles and Cayan 2002). This would in turn result in an approximate 20-percent reduction in historical annual spring runoff, and associated increases in winter flood peaks for the Sacramento/San Joaquin watershed (Knowles and Cayan 2002). With a doubling of atmospheric carbon dioxide from 280 to 560 parts per million, Snyder et al. (2002) project an 82-percent decrease in snow accumulation by the end of February in the central Sierra Nevada. Using two new climate models with different emission scenarios, Hayhoe et al. (2004) project a 50 to 75-percent and a 73 to 90-percent reduction in Sierra Nevada snowpack before 2100. A 60 to 70-percent reduction in snowpack in the Coast Ranges of California and Oregon is projected by 2040-2060 (Leung et al. 2004).

According to Maurer (2007), changes in precipitation, temperature, and snow water equivalence, will result in an earlier arrival of annual flow volume of 36 days in the Sierra Nevada by 2071-2100, with related decreases in spring and summer flows. As a result of less snowpack and earlier snowmelt, California can expect significantly less spring and summer runoff (Snyder et al. 2004) resulting in less water for ecosystem services, less reservoir capture, a diminished water supply for human uses, and greater conflict over the allocation of a diminished supply (Knowles and Cayan 2002, Kim et al. 2002, Snyder et al. 2004, Schlenker et al. 2007, Oregon Climate Change Research Institute 2010, Mayer and Naman 2011). A higher percentage of winter precipitation falling as rain rather than snow and the potential for more rain-on-snow events, also indicates greater flood frequencies during the cold season (Kim et al. 2002).

Aquatic ecosystems are likely to be impacted by these changes. Plants and animals that rely on snowmelt runoff or regular summer flows will experience streams and rivers with substantially lower summer flows or that goes dry much earlier in the year. Water temperatures are also likely to be warmer due to the decreased contribution of snowmelt (Carpenter et al. 1992, Snyder et al. 2004). In arid and semi-arid regions especially, riparian ecosystems are extremely sensitive to altered flow regimes and are likely to be degraded by diminished stream flows (Carpenter et al. 1992, Poff et al. 1997).

Extreme Precipitation and Flood Events

In a global study assessing the effects of a doubling of atmospheric carbon dioxide concentrations, Zwiers and Kharin (1998) project greater and more frequent extreme precipitation events almost everywhere, with a globally-averaged 20-year return event increasing by about one centimeters of rain per day, or less than 10 percent, and return periods for extreme precipitation events shortening by a factor of two. Regional climate projections indicate northern California is likely to experience an increase in the frequency and intensity of high precipitation and high runoff (extreme) storm events during the 21st Century (Bell et al. 2004, Kim 2005, Kim et al. 2002, Snyder et al. 2002, Kunkel 2003, Maurer 2007, California Energy Commission 2009c, USGCRP 2009, Mannshardt-Shamseldin et al. 2010, Dettinger 2011, Ralph and Dettinger 2011).

By projecting changes in streamflow output for eighteen stream gauging stations statewide, California Energy Commission scenarios show all California rivers studied will have an increase in average flow during January to April by the end of the century (years 2070–2099) compared to historical periods (California Energy Commission 2009c). Projections under certain high GHG scenarios predict spikes in February river flows of 60 percent above historic levels, and increases in December river flows of 20 percent to almost 40 percent in other scenarios (California Energy Commission 2009c).

Diffenbaugh (2005) projected an increase of up to 10 extreme precipitation events per year in the Pacific Northwest (up to a 140-percent increase) under a higher emission scenario with some variation depending on location within the region. While Bell et al. (2004) project the North Coast and North Lohontan (Modoc and Lassen Counties) basins average an additional 2.5 heavy rainfall events per year with a doubling of atmospheric carbon dioxide concentrations.

Future changes in the frequency and magnitude of extreme temperature and precipitation events could severely impact natural ecosystems such as wetland and riparian habitats (Anderegg et al. 2012), through changes in plant community composition and distribution, increased risk of species invasions and exotic diseases, and extinction (Diffenbaugh 2005). Based upon the scientific literature, climate change impacts on wetland and riparian habitats are anticipated to increase in the future, while simultaneously, the importance of these habitats to mitigate the impacts of climate change on fish and wildlife will become even more valuable.

Sea Level Rise

Sea level rise has enormous implications for coastal planning, land use, development, and the conservation of fish and wildlife habitat along California's 1,350 kilometer-long

(~840 miles) coast and is therefore of significant statewide concern (California Energy Commission 2006, 2009a, Executive Order S-13-08). The Intergovernmental Panel on Climate Change, IPCC (2007), estimates that global sea level rose an average of 1.7 ± 0.5 millimeters per year over the 20th century, based on tide gauge data from around the world and that sea level rose an average 3.1 ± 0.7 millimeters per year from 1993 to 2003, based upon precise satellite altimetry measurements. Tidal gauge data for California and the west coast of the United States have shown a similar trend in sea level rise (California Energy Commission 2006, 2009a).

Using future warming scenarios from IPCC (2007), Rahmstorf (2007) projected sea level rise by 2100 of 50 to 140 centimeters (~20 to 55 inches) above the 1990 level. A more recent analysis by Vermeer and Rahmstorf (2009), also using IPCC (2007) future warming scenarios, projects a sea level rise ranging from 75-190 centimeters (~30 to 75 inches) for the period 1990-2100. The most recent and detailed analysis of sea level rise for California's coast projects the following: south of Cape Mendocino, sea level will rise 4-30 centimeters by 2030 relative to 2000, 12-61 centimeters by 2050, and 42-167 centimeters by 2100; north of Cape Mendocino, sea level is projected to change between -4 centimeters (sea-level fall) and +23 centimeters by 2030, -3 centimeters and +48 centimeters by 2050, and 10-143 centimeters by 2100 (National Academy of Sciences 2012). The relative difference in sea level projected north and south of Cape Mendocino is primarily related to differences in coastal uplift rates due to plate tectonics. Changes in relative sea level rise are geographically variable because of local and regional differences in tectonic uplift; land subsidence; post-glacial isostatic rebound; compaction of sedimentary soils; oil, natural gas, and water withdrawal; and gravitational and deformational effects related to melting polar ice (USEPA 1988, Galbraith et al. 2002, Scavia et al. 2002, National Academy of Sciences 2012).

There is strong scientific consensus that coastal marine ecosystems, along with the goods and services they provide, are threatened by sea level rise (Church and Gregory 2001, Scavia et al. 2002, Harley et al. 2006, Nicholls and Tol 2006, California Energy Commission 2009a, USGCRP 2009). As a result of sea level rise, coastal areas worldwide are expected to experience higher rates of coastal erosion, flooding, inundation, and storm surges over the coming decades (USEPA 1988, Church and Gregory 2001, California Energy Commission 2006, 2009a, USGCRP 2009). According to Nicholls et al. (1999), by the 2080s, sea level rise could cause the loss of up to 22 percent of the world's coastal wetlands.

Increased sea levels, especially in combination with storm-driven surges, extreme waves, intense low-pressure autumn or winter storms, high tides, and El Niño conditions, are predicted to result in extensive flooding in coastal regions of California and the Pacific Northwest and significant damage to coastal infrastructure (California Energy Commission 2006, USGCRP 2009, National Academy of Sciences 2012). Kelvin waves, for instance, are generated in the tropical western Pacific during El Niño events and can intensify the impact of Northcoast winter storms. These waves move northward up the California coast bringing an influx of warm water and raising sea level by 15-25 centimeters (6-10 inches) as they pass (California Energy Commission 2006).

In an analysis of 140 years of tidal data from central California, Bromirski et al. (2003) found that since about 1950, California has experienced a significant increasing trend in extreme winter storms resulting in extreme high sea level residuals. Intense storm events cause the greatest coastal erosion and have the greatest impact on coastal development (Bromirski et al. 2003). In the past two decades, California has experienced significant increases in annual maximum wave heights and in the number of waves classified as extreme as a result of more intense El Niño events, though it is yet unclear if this trend is related to climate change (Seymour 2003).

Climate models predict the number of occasions when high sea levels and high river flows coincide will increase markedly in this century (California Energy Commission 2009b). According to California Energy Commission (2009b), “The combined impacts of sea level rise (and high sea-level stands) with concurrent river flood flows have the potential to imperil many smaller coastal and estuarine settings and communities along the California coast.” Sea level rise and related coastal erosion are expected to result in substantial losses of coastal wetland and intertidal habitats in the future (Nicholls et al. 1999, Galbraith et al. 2002, California Energy Commission 2006, 2009a).

A significant amount of coastline on the North Coast is located in the Erosion High Hazard Zone delineated on the California Flood Risk Sea Level Rise Maps and the built environments in these areas are threatened by sea level rise (California Climate Change Portal 2013). Because coastal wetlands are also particularly vulnerable to sea level rise-related inundation, flooding, and coastal erosion, undeveloped lands in or immediately adjacent to the coastal floodplain may be the only areas suitable for future wetland or estuarine habitat maintenance, restoration, and inland migration.

The California Energy Commission (2009a) includes the following sea level rise-related principles for adaptation and recommended practices and policies:

- Sea level rise must be integrated into the design of all coastal structures.
- Current efforts to build, maintain, or modify structures in coastal areas at risk of sea-level rise must be based on current estimates of projected rise.
- Development should be prohibited on land immediately adjacent to wetlands at risk of sea level rise. These buffer areas may be the only areas suitable for future wetland restoration projects.
- In areas at risk from sea level rise that are not heavily developed, local communities and coastal planning agencies have the opportunity to limit development and reduce future threats to life and property.

In summary, climate science indicates that in the coming decades, northern California is likely to experience warmer air and surface water temperatures, wetter winters, less snowpack and faster snowmelt, more frequent and severe drought, an increase in the frequency and severity of winter storms and flood events, and a rise in relative sea levels. Based upon the above, wetland and riparian habitat’s stream shading and cooling abilities, erosion-buffering properties, floodwater storage capacity, and groundwater recharge capabilities will be of even greater importance to fish and wildlife in the future. The scientific literature indicates that over the coming decades, it is highly likely climate change will magnify the already substantial adverse effects of land use and development on California's wetland and riparian habitats, even as their ecosystems services become more valuable. For these reasons, effective land use planning and project impact analysis and mitigation should include an assessment of future climate change and sea level rise impacts, when appropriate.

BUFFER EFFECTIVENESS

In its recommendations regarding the California Fish and Game Commission's Wetland Resources Policy, the Department stated that wetlands and associated uplands complement one another and that numerous animals found in wetlands are, nevertheless, at least partially dependent upon associated uplands. The Department recommended that buffers between proposed development and aquatic habitats should be included as an integral component of all mitigation plans for project impacts. The Department concluded that a "(f)ailure to retain this ecological bond between wetland and associated uplands will result in the creation of isolated wetland enclaves scatted throughout highly urbanized areas and result in indirect loss of wetland habitat values."

Riparian buffer or reserve guidelines developed by various jurisdictions reflect a diversity of management objectives, including the protection of water quality and wildlife habitat (Richardson et al. 2005). Riparian and wetland vegetation improves stream and wetland water quality by removing sediment, organic and inorganic nutrients, and toxic materials (Belt and O'Laughlin 1994, Mitsch and Gosselink 2000, USDA 2000, Meyer et al. 2006). Riparian buffers help keep pollutants from entering adjacent waters through a combination of processes including dilution, sequestration by plants and microbes, biodegradation, chemical degradation, volatilization, and entrapment within soil particles. The scientific literature shows effective wetland, stream and riparian buffers, in combination with other conservation strategies, may help to avoid and mitigate for the land use and development impacts described in the preceding sections of this Technical Memorandum. Site specific conditions may justify wider or narrower, or variable buffer widths. Such circumstances may include, for example, the presence of State or federally-listed species, SSC or especially sensitive or significant habitats, such as coastal lagoons or vernal pool complexes. Special consideration should be given to vernal pool buffer widths because vernal pool hydroperiods are acutely driven by the characteristics of surrounding uplands such as soil characteristics, gradient, size and configuration of the uplands, and potential hydrologic project impacts.

While no set habitat buffer width or mitigation strategy can be shown to be effective or necessary in all instances, the most recent and best available science provides technical guidance on buffer width and other buffer characteristics that are likely to be most effective on a landscape-scale. For example, road construction and forest removal on surrounding lands was shown to significantly affect biodiversity in adjacent wetlands (Findlay and Houlahan 1997). Their data suggest wetland policies that only protect the wetland or a narrow buffer zone around its perimeter, "…are unlikely to provide adequate protection for wetland biodiversity (Findlay and Houlahan 1997)."

There is substantial evidence showing narrow buffers are considerably less effective in minimizing the effects of adjacent development than wider buffers (Castelle et al. 1992, Brosofske et al. 1997, Dong et al. 1998, Kiffney et al. 2003, Moore et al. 2005). Hilty and Merelender (2002), for instance, studied the use of stream corridors by predatory mammals in Sonoma Co. California, and found habitat use varied greatly by riparian corridor width. They sampled three riparian corridors types: 1) denuded corridors had very little natural vegetation along the creek; 2) narrow corridors had a strip of vegetation ranging from 10 to 30 meters on each side of the creek; and 3) wide corridors had more than 30 meters of natural vegetation on each side of the creek. Key results of their study include: 1) "Significantly more species of mammal predators were detected in wide riparian corridor sites than narrow or denuded sites;" 2) "A greater diversity of all mammalian predators and more native mammal predators were found in wide riparian corridors, compared to narrow or denuded corridors;" 3) "Large native predators were detected primarily in wide riparian corridors, and smaller native and non-native mammalian predators, especially the domestic cat, were more active in narrow and denuded riparian corridors" (Hilty and Merelender 2002).

Substantial research conducted in diverse riparian habitats across North America has shown that buffers of at least 50 to 100 meters wide (164 to 328 feet) are required to maintain avian biodiversity (Fischer and Fischenich 2000). Friesen et al. (1995) found that Neotropical songbirds consistently decreased in diversity and abundance as the level of adjacent development increased. Their study showed 10 acre (four hectare) woodlots without any nearby houses had on average a richer, more abundant Neotropical bird community than 61 acre (25-hectare) urban woodlots. On streams in southeastern British Columbia, Canada, Kinley and Newhouse (1997) determined that riparian reserves averaging 70 meters wide (230 feet) are needed to support near-natural densities of riparian-associated birds. In a study of boreal mixed-wood forest in Alberta, Canada, Hannon et al. (2002) found forest-dependent bird species declined as riparian buffer widths narrowed from 200 to 100 meters (656 to 328 feet). They found 20-100 meter (65-328 feet) riparian buffers would not conserve forest songbird populations, but 200 meter-wide (628 feet) buffers would maintain pre-timber harvest passerine bird communities (Hannon et al. 2002)

Numerous other studies document how human activities in natural areas disturb bird populations and reduce bird diversity, abundance, and reproductive success (Burger et al. 2004, Banks and Bryant 2007, Fernandez-Juricic 2002, Rodgers and Smith 1997). According to the USACE technical note, "If avian habitat is a management objective, managers should consider managing for riparian zones that are at least 328 feet (100 meters) wide." (Fischer 2000). This USACE recommendation applies to either side of the channel in larger river systems and to total width for lower-order streams and rivers (Fischer 2000).

Because of the impacts of edge effects on riparian habitat, Crawford and Semlitsch (2007) recommend an overall buffer of 97 meters (318 feet) for southern Appalachian salamander streams. Harper et al. (2008) determined that regulations protecting 30 meters (98 feet) or less of surrounding terrestrial habitat are inadequate to support viable populations of pool-breeding amphibians. Olson et al. (2007) recommend a conservation approach that utilizes riparian management zone buffers of 40 to 150 meters wide (131 to 492 feet) on headwater streams to accommodate terrestrial life history functions of riparian associated fauna.

According to the USEPA, riparian buffers are a best management practice that should be used in conjunction with comprehensive watershed management plans to control and reduce point and non-point sources of nitrogen into the nation's aquatic habitats (Mayer et al. 2006). As buffer width increases, the effectiveness of removing pollutants from surface water runoff increases (Castelle et al. 1992). To protect water quality in Oklahoma's streams, a 29 meter-wide (95-feet) riparian buffer is recommended by the Oklahoma Cooperative Extension Service (Harmel et al. undated). By using benthic macroinvertebrate levels and salmonid egg development, studies generally found 30 meter (approximately 100 feet) buffers were effective in preventing water quality impacts from stormwater runoff (Castelle et al. 1992). For Georgia streams, a 30 meter (approximately 100 feet) wide riparian buffer is considered sufficiently wide to trap sediments under most circumstances, although it is recommended buffers should be extended for steeper slopes (Wenger and Fowler 2000).

Effective buffers also minimize disturbance to wetland and riparian habitats by limiting human access, minimizing refuse dumping and invasive species introductions, and by blocking transmittal of light and noise. In an analysis of wetland buffer width effectiveness on 100 coastal wetland sites in New Jersey, Shisler et al. (1987) found disturbance levels (dumping, vegetation removal, illegal lot build-out, etc.) were double at sites with narrow buffers, less than 50-feet (15 meters), than buffers 30 meters (approximately 100 feet) wide or greater. Buffers of 100-feet and greater (>30 meters) provided significantly more protection and lower levels of disturbance than buffers less than 50-feet wide (Shisler et al. 1987, Castelle et al. 1992). This study recommends buffer widths of 100, 100, and 150-feet (approximately 30, 30, and 45 meters), respectively for salt marshes, hardwood swamps, and tidal freshwater marshes for high intensity land uses such as high-density residential and industrial/commercial development (Shisler et al.1987, Castelle et al. 1992). According to the Natural Resources Conservation Service, "…a minimum buffer of 30 meters (approximately 100 feet) on both sides of the stream is recommended for sufficient stream protection. This usually amounts to a buffer that is 3-5 mature trees wide on each side of the stream" (Natural Resource Conservation Service 2004).

From a literature review conducted by Castelle et al. (1994), it appears buffers of less than 5 to 10 meters (16 to 33 feet) provide little protection of aquatic resources under most conditions. Based upon their analysis, buffers necessary to protect wetlands and streams should be 15 to 30 meters (approximately 50 to 100 feet) in width under most circumstances (Castelle et al. 1994). However, these authors note that site-specific conditions may indicate a need for substantially larger buffers or somewhat smaller buffers.

Brosofske et al. (1997) studied the effect timber harvesting had on the microclimate of adjacent riparian forests in western Washington State. Their research concluded that a buffer at least 45 meters wide (approximately 150 feet) on each side of a stream is necessary to maintain a natural riparian microclimate along the streams they studied, which were characterized as having a 70 to 80-percent overstory canopy of predominantly conifers, and a regional climate typified by hot, dry summers and mild, wet winters (Brosofske et al. 1997). However, these researchers also found, depending on the microclimate variable, buffer widths of up to 300 meters (984 feet) may be needed to maintain an unaltered microclimate (see Chen et al. 1999).

In California's Coastal Zone, development buffers on streams, wetlands, and other environmentally sensitive habitat areas are determined by local coastal plans (LCPs). The most common buffer dimension required in city and county LCPs is 100 feet (California Coastal Commission 2007). According to a report by the California Coastal Commission, the majority of LCPs state a 100-foot (30 meter) buffer is the minimum standard, and especially sensitive habitats may require a larger buffer. Despite this, the report found that across the state, the width of currently applied LCP buffers fall short of buffer dimensions shown to be effective by the scientific literature (California Coastal Commission 2007). The Coastal Commission's analysis showed 30 to 59 meter-wide (approximately 100 to 195 feet) riparian buffers are generally accepted in the scientific literature as effectively protecting aquatic resources (California Coastal Commission 2007).

The scientific literature indicates an appropriate wetland or riparian buffer width would depend upon a number of site-specific characteristics, including: the area and type of habitat being buffered; presence of habitat for sensitive species and their potential habitat use (e.g. breeding vs. foraging or resting); sensitivity of the habitat or target wildlife species to disturbance; site topography, slope, slope stability, and soils; the habitat's rarity, quality, and connectivity or isolation from other natural communities; the habitat's potential for restoration; and the potential direct and indirect impacts from proposed adjacent development or other land use.

Utilizing the California Rapid Assessment Method (CRAM) is one tool that may help evaluate some of these habitat characteristics (California Wetlands Monitoring Workgroup 2009). However, given the diversity of wetland and riparian-dependent species and variability in potential direct and indirect impacts of development and land uses, numerous science-based buffer widths have been recommended to protect and maintain water quality and wildlife habitat.

For instance, according to a summary of the scientific literature by Fischer et al. (2000), buffer widths to protect and maintain the following are recommended: water quality (≥ 4 meters to ≥ 30 meters) (12 to 98 feet); reptile/amphibian habitat (≥ 30 meters and up to 1,000 meters) (98 to 3280 feet); bird habitat (> 40 meters and up to 1,600 meters) (131 to 5,250 feet); mammal habitat (≥ 50 meters) (164 feet); and plant diversity (≥ 30 meters) (98 feet). An analysis of 65 wetland and riparian-dependent amphibian and reptile species showed their core upland habitats ranged more than 100 meters (328 feet) from aquatic habitats, indicating that buffers would need to be at least that wide to effectively protect wetland and riparian-dependent amphibians and reptiles (Semlitsch and Bodie 2003). According to Fischer and Fischenich (2000), buffer widths of 100 meters (328 feet) or more are usually needed to ensure protection of wildlife habitat values and use as migration corridors; while increasing widths to encompass the geomorphic floodplain is likewise desirable to optimize flood-reduction benefits.

In summary, wetland and riparian buffers, in combination with other conservation strategies, can effectively avoid or mitigate development and land use impacts on wetland, stream and riparian habitats. While no set habitat buffer width or mitigation strategy can be shown to be effective or necessary in all instances, the most recent and best available science provides technical guidance on buffer width and other buffer characteristics that are likely to be most effective on a site-specific or landscape-scale. The scientific literature indicates that to maintain viable habitat for many of California's riparian and wetland dependent bird, amphibian, and reptile populations, an undeveloped upland habitat buffer of at least 50 meters wide (164 feet), and often considerably wider, would likely be necessary. The appropriate buffer width for a project should be based on project-specific direct and indirect impacts and habitat needs.

KEY FINDINGS

The following key findings are based on the preceding review of the scientific literature related to development, land use, and climate-change-related impacts on wetland and riparian resources. These findings highlight issues to consider when developing or reviewing individual projects or land use plans, but they are not to be relied on as a replacement for project or site-specific review and analysis.

1) The Governor of California and the Fish and Game Commission have developed policies seeking no overall net loss and long-term net gain in the quantity, quality, and permanence of wetlands acreage and values in California.

2) California's wetland and riparian habitats and the fish and wildlife they support are valuable and finite resources that benefit the people of the State and are threatened with loss and degradation. Consequently, the public interest requires coordinated efforts to preserve these natural resources and the ecological, recreational and economic benefits they provide.

3) Effective regional land use planning can be one of the best means to protect and restore California's remaining wetland and riparian habitats.

4) Wetland and riparian buffers, in combination with other conservation strategies, can effectively avoid or mitigate development and land use impacts on wetland, stream and riparian habitats. While no set habitat buffer width or mitigation strategy can be shown to be effective or necessary in all instances, the most recent and best available science provides technical guidance on buffer width

and other buffer characteristics that are likely to be most effective on a site-specific or landscape-scale. The scientific literature indicates that to maintain viable habitat for many of California's riparian and wetland dependent bird, amphibian, and reptile populations, an undeveloped upland habitat buffer of at least 50 meters wide (164 feet), and often considerably wider, would likely be necessary. The appropriate buffer width for a project should be based on project-specific direct and indirect impacts and habitat needs.

5) To most effectively protect wetland and riparian habitats in flood-prone areas and to avoid or decrease the risk of inundation, erosion, and flood damage to development, floodplains not already protected by levees are best managed for natural riverine processes such as floodwater storage and channel migration.

6) The scientific literature anticipates climate change and sea level rise will worsen many of the current threats to wetland and riparian resources. In addition, tools for modeling climate change, including changes in precipitation patterns, flood frequency and magnitude, and drought and wildfire patterns are evolving. Therefore, utilizing the most recent and best available climate science data will enable lead agencies and the public to most effectively and accurately evaluate the future impacts and implications of climate change and sea level rise on a project or land use plan.

7) Low-impact development techniques to slow, spread, and infiltrate stormwater on-site, rather than a more traditional approach to store, concentrate and drain stormwater off-site as quickly as possible, can benefit wetland and riparian habitats in three ways: 1) filtering out pollution and increasing the quality of stormwater runoff; 2) decreasing peak flows, flood risks and erosion in downstream waters; and 3) increasing ground water recharge and therefore helping maintain biologically-important summer low flows.

8) There is strong scientific evidence that requiring domestic housecats to be kept indoors at residential developments adjacent to wetland and riparian habitats could minimize the substantial impacts of outdoor housecats on wildlife populations. This can be enforced through covenants, conditions and restrictions enforceable by homeowner's groups or through general plan policies, zoning or land use ordinances.

9) Utilizing exterior light fixtures and street standards that are fully-shielded and designed and installed to minimize off-site glare and photo-pollution into wetland and riparian habitats and buffer areas can be an effective mitigation measure to minimize the impacts of artificial night lighting.

Conservation science is an ever-changing field. The scientific understanding of species, habitats, and threats changes over time as new research is conducted. The rarity, abundance, distribution, vulnerability, and listing status of species also change over time. Local, state, and federal regulations are not static. Land use and landscape characteristics change over time due to drought, flood, wildfire, development patterns, restoration efforts, and climate change. For these reasons, this Technical Memorandum should not be considered the definitive science on this subject. Stakeholders involved in assessing and avoiding impacts on wetland and riparian resources should rely on the best available science, baseline conditions, current law and policy, and the restoration potential of the site.

Commonly Used Methods for Implementing Wetland and Riparian Habitat Buffers

Habitat buffers require certain implementation techniques or methods to guide their design and ensure effectiveness. Numerous effective habitat buffer implementation methods exist. As discussed above, environmental impact analysis and mitigation require site-specific analysis and consideration, and the design and criteria appropriate for a specific buffer should be based on site-specific analysis and circumstances. The Department has worked with many stakeholders to implement habitat buffers. Below are a number of commonly used buffer implementation methods that the Department is familiar with and has considered effective:

1) Riparian habitat buffers begin at the outer edge (drip-line) of riparian canopy, if present, or top of stream bank, if riparian canopy is absent.

2) Wetland buffers begin at the edge of the delineated wetland.

3) Habitat buffers are measured using horizontal distance, perpendicular to the stream or wetland, regardless of slope.

4) Habitat buffers are applied to both left and right banks of streams and rivers.

5) Habitat buffers are considered undeveloped, no-disturbance areas. Hardscape such as structures and parking areas, septic systems, and stormwater treatment facilities are situated outside of habitat buffers. Exceptions may include trails for non-motorized use.

6) Where project construction necessitates temporary ground disturbance and vegetation removal in the habitat buffer, the disturbed buffer area should be restored to enhance fish and wildlife habitats and water quality. This enhancement could include decompacting soil, site recontouring, and revegetation with native species of local genetic stock.

7) Habitat buffers are graphically shown on project drawings and subdivision maps submitted for lead and permitting agency approval and subsequent recordation.

8) Habitat buffers are legally described and recorded on the appropriate Assessor's Parcel Maps. At the request of the property owner, a local agency may accept an offer of dedication and accept fee title to the habitat buffer area.

9) Habitat buffers are clearly marked and barrier fences are installed in the field during construction activities to prevent impacts from equipment operations.

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