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Mountain-to-sea ecological-resource management: forested watersheds, coastal aquifers, and groundwater dependent ecosystems

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Highlights

- Optimal investment in watershed conservation capital is usually positive (non-zero)
- Upstream watershed conservation supports protection of downstream GDEs
- Stricter safe minimum standards for the GDE necessitate more investment
- Optimal investment is lower when capital is more efficient at capturing recharge
- Growing water demand requires more investment in the optimal solution

Abstract

Improving the understanding of connections spanning from mountain to sea and integrating those connections into decision models have been increasingly recognized as key to effective coastal resource management. In this paper, we aim to improve our understanding of the relative importance of linkages between a forested watershed, a coastal groundwater aquifer, and a nearshore marine groundwater-dependent ecosystem (GDE) using a dynamic groundwater optimization framework and simple ecosystem equations. Data from the Kīholo aquifer on the Kona Coast of Hawaiʻi Island are used to numerically illustrate optimal joint management strategies and test the sensitivity of those strategies to variations in physical and behavioral parameter values. We find that for a plausible range of watershed management costs, protecting part of the recharge capture area is always optimal. Without watershed protection, maintaining a safe minimum standard growth rate for a GDE-dependent marine indicator species, reduces net present value non-trivially, but optimal investment in watershed conservation offsets that potential reduction by 75%. In general, we find that optimal watershed management and groundwater pumping are most sensitive to changes in water demand growth and parameters that describe nearshore salinity.

Keywords: groundwater management; watershed conservation; groundwater dependent ecosystems; dynamic optimization

Introduction

In recent decades, research on ecosystem services, the benefits ecosystems provide to people, has continued to grow. Much effort has already been expended to develop a unifying framework and systematic methodology for including ecosystem services in resource and land-use planning (Chan et al., 2006; Daily and Matson, 2008; Brauman et al., 2007; Daily et al., 2009), and research on specific subsets of services such as watershed hydrologic services (Lele, 2009; Asbjornsen et al., 2015) have continued to advance as well. However, as Brauman (2015, p. 345) points out, many contributions to the literature “use the language of hydrologic services but appear to be essentially disciplinary studies, accounting for either biophysical functioning or specific beneficiaries in their analysis, but not both.” Lele (2009) further draws attention to both the need for better scientific understanding of biophysical linkages in many regions and improved integration of biophysical models in economic valuation studies. Challenges of integration notwithstanding, research on ecosystem services in economic theory and practice has continued to forge ahead (Engel et al., 2008; Gómez-Baggethun et al., 2010), with recent efforts largely trending toward the design and implementation of payment for ecosystem services (PES) schemes (Salzman et al., 2018). Our aim in this paper is to develop an optimal control model for water resources management that incorporates ecosystem equations and improves our understanding of a complex hydrological-ecological-economic problem from a systems perspective. While the ecosystem equations are overly simplistic to ensure tractability, we believe this approach is an important step toward addressing some of the gaps described by Brauman (2015) and Lele (2009). In particular, we highlight the linkages between maintenance of an upstream high-elevation watershed, a coastal aquifer, and a downstream marine ecosystem. The groundwater aquifer is the connecting resource through which the upstream benefits are conferred to the downstream coastal ecosystem, but the methodology can be generalized to other cases in which improvement of an upstream resource confers benefits to a downstream resource via a mutually linked intermediate resource. Other linked systems might include connections such as those between upstream forest structure, water quality, and related fish/marine populations, or connections between upstream soil stock/health, downstream sedimentation rates, and coral reef and linked marine organism abundances.

Although hydrologic services such as groundwater recharge are typically viewed as being provided to the freshwater resource from an upstream ecosystem, the water resource itself can also sometimes provide a service to a linked ecosystem downstream. One such example is groundwater-dependent ecosystems (GDE). The importance of better understanding GDEs has become widely recognized (Brown et al., 2010; Moosdorf and Oehler, 2017; Tomlinson and Boulton, 2010), and the protection of GDEs has been incorporated into a number of water policy initiatives in Australia, the European Union, South Africa, and the

United States (Rohde et al., 2017). With a few exceptions (e.g., Pongkijvorasin et al., 2010), however, less attention has been paid to the potential value of including GDEs in economic groundwater optimization frameworks. In combining hydrologic ecosystem service provision and maintenance of a nearshore marine GDE with a coastal groundwater optimization model, our approach allows us to better understand the relative importance of different types of system linkages that drive dynamic mountain-to-sea water management outcomes.

Linkages from “mountain to sea” or “ridge to reef” have been recognized as key components of integrated coastal management. However, quantitative evidence of these relationships is often lacking (Rodgers et al., 2012), and planning decisions are regularly made under data-limited conditions (Rude et al., 2015). Recent research in this area has aimed to improve our understanding of these linkages (Delevaux et al., 2018) but such studies typically do not build those relationships directly into an integrated management framework. Instead, results from the biophysical model are presented as the final output, and resource managers are tasked with translating those results into real world management decisions. In this paper, we begin to bridge that “mountain to sea” gap by incorporating linkages between a forested watershed, a coastal groundwater aquifer, and a nearshore marine ecosystem into a dynamic optimization framework. Data from the water-limited Kīholo aquifer region on the Kona Coast of Hawaiʻi Island are used to numerically illustrate optimal joint management strategies and test the sensitivity of those strategies to variations in a number of physical and behavioral parameter values. We then use those results to address the following research question: To what extent can investment in watershed conservation be used as a tool to reduce the costs to groundwater users of satisfying a nearshore ecological constraint? While results are specific to this case study, our “mountain to sea” framework is generalizable to other linked systems where management of one (often upstream) system has influence on another (often downstream) system via an intermediate resource with physical, ecological, and/or economic linkages to both.

Methods

Given our objective to better understand the potential of watershed conservation as a management tool to reduce the cost to groundwater users of satisfying a nearshore marine ecological constraint, the modeling framework requires the integration of three major components: (i) the recharge capture zone in which watershed conservation occurs, (ii) the groundwater resource which receives inputs from (i) and is pumped for human use, and (iii) the nearshore ecosystem whose health depends on the inflow it receives from (ii). Fig. 1 illustrates the complete mountain-to-sea system.

Figure 1. Mountain to sea management of a forested watershed, coastal aquifer, and groundwater-dependent ecosystem.

The groundwater resource is the centerpiece of the system, as it is connected directly to all of the other components. In order to examine consequences to groundwater management while considering these interlinked resources, we follow Krulce et al. (1997) and Duarte et al. (2010) and model this piece as a single-cell coastal aquifer with a sharp freshwater-seawater interface, where the head level (h), or the distance between mean sea level and the top of the groundwater lens, is proportional to the volume of stored freshwater. In general, this type of simple relationship will not adequately capture the complexity of density-dependent flow dynamics and geology in a coastal aquifer. However, it is sufficient for tracking average volumes of water and head level in the aquifer as required for our analysis. The head level changes over time in accordance with recharge (*R*), leakage or discharge at the coast (*l*), and extraction (*q*):

$$\dot{h}_t = \gamma[R(N_t) - l(h_t) - q_t] \tag{1}$$

where $\dot{h}_t$ denotes the derivative of *h* with respect to time and γ is a volume-height conversion factor. Leakage is an increasing and convex function of the head level, i.e. $l'(h_t) > 0$ and $l''(h_t) \leq 0$. When the head level is higher, more pressure along the aquifer boundary generates more leakage in the form of submarine groundwater discharge.

As indicated by Eq. 1, recharge to the aquifer is a function of the watershed conservation capital stock (*N*). Assuming that rainfall is constant and exogenous, *R* is constrained from above by the rate corresponding to the maximum feasible conservation capital stock (N^{max}). We also assume that the recharge function is increasing and convex in the capital stock, i.e. $R'(N_t) > 0$ and $R''(N_t) \leq 0$. That is, adding to the capital stock contributes to recharge but the marginal benefit is non-increasing. The capital stock evolves dynamically based on investment decisions (*I*) and depreciation (δ):

$$\dot{N}_t = I_t - \delta N_t \tag{2}$$

From Eq. 2, we can see that investment decisions drive the path of the conservation capital stock over time, which then feeds into the aquifer state equation via the recharge function.

The third piece of the integrated system is the nearshore groundwater-dependent resource. Depending on the particular application, this resource may be harvested and exchanged at a market price, in which case the surplus generated by such transactions could be included directly in the objective functional for the maximization problem. Often times, however, interest in protecting linked ecosystems as part of an integrated water management strategy is motivated by nonmarket values (e.g., biodiversity and habitat protection, cultural and traditional uses, aesthetics), which are typically difficult to quantify in monetary terms. In that case, a safe minimum standard approach may be more appropriate, wherein an ecological parameter or set of parameters are constrained within the model. For our system, following Duarte et al. (2010), we require that the growth rate of an indicator species (*g*) be maintained at or above a target rate (*m*):

$$g\big(l(h_t)\big) \geq m \tag{3}$$

where $\frac{\partial g}{\partial h_t} > 0$. Eq. 3 illustrates the link between the aquifer system and the nearshore marine ecosystem. Investment in recharge-enhancing capital stock and extraction from the aquifer affect future head levels, which subsequently has an impact on the growth rate of the downstream indicator species.

With the state equations and constraints in place (Eq. 1-3), we next define the benefit and cost functions that will be used in the objective functional of the dynamic optimization problem. We measure the benefit of groundwater use as the area under the inverse demand curve (D^{-1}):

$$B_t = \int_0^{q_t+b_t} D^{-1}(x_t, t)\, dx \tag{4}$$

where groundwater extraction (q) can be supplemented by a backstop resource (b) such as desalination if it is optimal to do so.[1] We assume that the sources are indistinguishable to users and therefore generate equivalent marginal benefit. Demand is also a function of time (t) to allow for the possibility of exogenous water demand growth. The unit costs of desalination (c_b) and investment in conservation capital (c_I) are assumed constant and exogenous, but the marginal cost of groundwater extraction (c_q) is a decreasing and convex function of head, i.e. $c_q{}'(h_t) < 0$ and $c_q{}''(h_t) \geq 0$. As the head level is drawn down, more energy is required to lift groundwater over a longer distance to the surface. The total cost is summarized as follows:

$$C_t = c_q(h_t)q_t + c_b b_t + c_I I_t \tag{5}$$

The resource manager must then choose the rates of groundwater extraction (q), desalination (b), and investment (I) in conservation capital stock in every period, given a discount rate $r > 0$, to maximize the net present value of social welfare:

$$\max_{q_t, b_t, I_t} \int_0^\infty e^{-rt}\{B_t - C_t\}\, dt \tag{6}$$

subject to Eq. 1-3 and non-negativity constraints on the control and state variables. For further interpretation of the tradeoffs involved in the optimal solution, we can rewrite Eq. 6 as a current value Hamiltonian:

$$\mathrm{H} = B_t - C_t + \gamma\lambda_t[R(N_t) - l(h_t) - q_t] + \mu_t[I_t - \delta N_t] + \theta_t\left[g\big(l(h_t)\big) - m\right] \tag{7}$$

[1] To avoid confusion that may arise by writing the sum of q and b as both the upper limit of integration and the variable of integration, x plays the role of a dummy variable in Eq. 4. That is, the integral represents the area under the inverse demand curve from 0 up to the total quantity of water consumed (the sum of q and b) in period t.

where λ_t and μ_t are the costate variables for the aquifer and conservation capital stocks respectively, and θ_t is the shadow price of the ecological constraint. From the necessary conditions for Eq. 7, one can derive the following expression for the efficiency price of groundwater (i.e., the price that would incentivize the optimal rate of water extraction):

$$p_t = c_q(h_t) + \frac{\dot{p}_t - \gamma\{c_q{}'(h_t)\left[R(N_t) - l(h_t) + \theta_t \frac{\partial g}{\partial l} l'(h_t)\right]\}}{r + \gamma l'(h_t)} \tag{8}$$

where $p_t \equiv D^{-1}(q_t + b_t)$. The second term on the right-hand side of Eq. 8 is the marginal user cost (MUC) of groundwater extraction. The last term in the numerator of the MUC captures the external cost that extraction has on the growth of algae, while the second term allows for the effect of conservation capital on the aquifer state. Similarly, one can use the necessary conditions of Eq. 7 to derive an optimality condition for capital investment:

$$\lambda_t R'(N_t) = c_I(r + \delta) \tag{9}$$

which suggests that at the optimum, the marginal benefit of investment, valued at the shadow price of groundwater, should be equal to the marginal cost, which includes foregone interest and the cost of depreciation. Eq. 9 tells us that the optimal rate of investment depends on the scarcity of the linked groundwater resource. In the numerical application that follows, the optimization simulation is run to solve Eq. 6.

Application

Our study site on the North Kona coast of Hawai‘i Island overlies a thin basal groundwater lens, known locally as the Kīholo aquifer. Due to the high porosity (ratio of pore space available for transmission of fluids to the total volume of rock) in Kīholo, the brackish transition zone at the interface between fresh groundwater and underlying saltwater is relatively thin (Duarte, 2002). Accordingly, we follow others in the groundwater economics literature (Krulce et al., 1997; Koundouri and Christou, 2006; Roumasset and Wada, 2012) and approximate aquifer storage and dynamics under the assumption that head is proportional to stored volume in every period; in particular, we assume that a sharp interface is located below mean sea level at a depth of approximately 40 times the head level (Mink, 1980). Following Pongkijvorasin et al. (2010), the equation of motion for the head level is expressed as $\dot{h}_t = \gamma(R(N_t) - l(h_t) - q_t)$, where $\gamma = \left(\frac{2000}{41\theta WL}\right)$, θ is porosity, and W and L are aquifer width and length in meters respectively. Plugging in parameter values $\theta = 0.3$, $W = 6000$, and $L = 6850$ for Kīholo (Pongkijvorasin et al., 2010) yields a volume-to-head conversion factor $\gamma = 0.00000396$ m per thousand m^3, or equivalently 0.000049 ft per MG (million gallons). We believe that the relatively simple characterization of the aquifer is sufficient for our purposes given the relatively calm sea conditions, homogeneous coastline geology, lack of surface water inputs or outputs, and unidirectional flow to the ocean along the coastline at our study site (Duarte et al., 2010). However, because the actual fresh-saltwater interface is not completely sharp in reality, we test the sensitivity of our results to variations in the value of γ (see the appendix).

The current rate of recharge (R) for Kīholo aquifer is assumed to be 3992.7 MG per year (Pongkijvorasin et al., 2010). However, our model allows this value to increase in the future if investment in watershed management activities, such as the construction and maintenance of ungulate-proof fencing, are undertaken. Browsing and trampling by non-native feral ungulates in Hawaiʻi have been observed to negatively impact groundcover and cause soil compaction, which ultimately affect runoff and infiltration rates (Dunkell et al., 2011). Preventing incursion of ungulates into upstream forested watershed areas, therefore, supports higher recharge rates. Wada et al. (2019) found that current investment in watershed conservation across several management sites on Hawaiʻi Island generate additional recharge in the range of 0.0012-0.1712 MG per acre annually when averaged over a 50-year period. We assume that the marginal recharge gained per dollar invested in fencing decreases linearly over that range across the roughly 10,000-acre recharge capture zone directly above Kīholo aquifer. In other words, the first protected acre adds 0.1712 MG/yr, while the 10,000th protected acre adds 0.0012 MG/yr, and recharge in any given year is calculated using the following equation in the baseline analysis: $R(N_t) = 3992.7 + 0.172N_t - 0.000008N_t^2$, where *N* represents the number of fenced acres. Given the uncertainty regarding the recharge returns to investment in forest protection, we test the sensitivity of our results to the values of the recharge function coefficients.

Outflow from the aquifer occurs either naturally as leakage/discharge (*l*) along the coast or as pumping from groundwater wells (*q*). When the interface between fresh and saltwater is assumed to be sharp, Mink (1980) shows that the structural expression for leakage is $l(h_t) = zh_t^2$. Given the current rate of groundwater extraction and head level, Pongkijvorasin et al. (2010) estimate that the value of parameter z is between 113 and 121 for for Kīholo aquifer when leakage is measured in MG and head is measured in ft. For our baseline analysis, we assume z = 117, but we also test the sensitivity of our model results to this assumption (see the appendix).

Pumped groundwater, the primary source of freshwater in the region, supplies municipal users whose consumption preferences are characterized by the following inverse demand function: $p_t = 15.066 - 0.0086 q_t e^{-\alpha t}$, where α is the annual growth rate of demand and p is the marginal benefit of water in USD per MG (Pongkijvorasin et al., 2018). In the baseline case, we assume that $\alpha = 0$, but we consider demand growth rates of 0.25, 0.5, and 1.0 percent in the sensitivity analysis. The coefficients for the demand function were determined using the local water utility's rate structure and an assumed price elasticity of -0.7 (Griffin 2006; Dalhuisen et al., 2003). We use the demand curve to estimate consumer surplus and subsequently the net present value (NPV) associated with each of the simulation optimization solutions.

The optimal (NPV-maximizing) solution is also dependent on a number of different costs. The cost of pumping groundwater is driven primarily by the cost of energy required to lift water to ground level. Following Pongkijvorasin et al. (2018), we assume that the marginal cost of water extraction is a linear function of lift: $c(h_t) = 0.00115(e - h_t)$, where $e = 1322.82$ ft is the average ground surface elevation in Kīholo and c is measured in USD/MG.

In the case that the optimal solution calls for implementation of a backstop resource, we further assume that desalinated seawater can be produced at a constant unit cost of \$7,570/MG, inclusive of amortized capital costs (Pongkijvorasin et al., 2018). The cost of watershed management is driven primarily by fence installation and maintenance costs. Assuming that the fenced area is square, each acre requires approximately 835 ft of fence at an installation cost of \$30/ft (Wada et al., 2019). Based on current management practices on Hawaiʻi Island, we also assume that the choice to install obligates maintenance in two intervals:

(i) \$4.68-8.72 per ft every 5 years for fence wire replacement, (ii) \$20 per ft every 30 years for wire and post replacement (Wada et al., 2019). Summing and discounting the trajectory of installation and maintenance costs over 50 years results in a per unit present value cost (c_f) in the range of \$49-82/ft.[2] In the baseline simulation, we assume a cost of \$75/ft. Note that because we are assuming depreciation-offsetting maintenance is already included in the present value cost of the fence, the equation of motion for the conservation capital stock (Eq. 2) simplifies to $\dot{N}_t = I_t$.

The final component of the mountain-to-sea system is the nearshore marine resource. Following Duarte et al. (2010), we focus on the Hawaiian endemic algae, limu manauea (*Gracilaria coronopifolia*), which is ecologically and culturally significant in Hawaiʻi. Lab experiments have shown that the growth of this keystone indicator species varies with the salinity of its habitat according to the following relationship: $g_t = 10.2975 - 0.2625 s_t$, where *g* is the growth rate of limu manauea and *s* is salinity in parts per thousand (ppt) (Duarte et al., 2010). The level of salinity, in turn, is a function of the rate of submarine groundwater discharge along the coast. Given that the average nearshore coastal salinity at the study site is ~31 ppt, and assuming that nearshore salinity would be equal to that of seawater (36 ppt) in the absence of discharge, the relationship between salinity and discharge can be expressed linearly as follows: $s_t = 36 - 0.00125 l(h_t)$ (Duarte et al., 2010), where leakage is measured in MG. For the preliminary model runs, we consider minimum growth constraints of 1.8% and 2.0%. We

[2] We consider a range of fence costs because the labor requirements for installation and maintenance have historically varied across management units on Hawaiʻi Island, due primarily to differences in terrain and accessibility.

also consider the possibility that nearshore salinity varies more or less with discharge than in the baseline case (see the appendix). Key equations required for the numerical simulations are summarized in Table 1.

Table 1. Key equations summarizing the mountain-to-sea system

Results

We consider first the impact of imposing a minimum ecological growth constraint on optimal groundwater pumping from Kīholo aquifer. As expected, tightening the constraint shifts the optimal groundwater trajectory downward (Fig. 2). Net present value (NPV) over 50 years for the unconstrained optimum is $282.78 million. Enforcing a baseline growth constraint of 1.8% slightly reduces NPV to $271.21 million. Increasing the minimum required growth rate by 10% of the baseline rate (from 1.8% to 2.0%), however, results in a precipitous drop in NPV to $186.04 million, due primarily to early implementation of the costly backstop resource starting in year 8. If we impose no ecological constraint but allow for investment in fencing for watershed protection, the optimal solution would exclude fence construction. The resulting NPV of $282.78 million is the same as the unconstrained optimum described above. Building a fence to cover the entire recharge area reduces NPV by the cost of fence construction and maintenance to $276.54 million.

Figure 2. Groundwater extraction for the unconstrained optimum and two growth constraints on indicator species *Gracilaria coronopifolia* with no fencing.

With our baseline growth constraint of 1.8% imposed on the algae indicator species, the NPV can be increased through optimal investment in fencing. Fully fencing the recharge area increases NPV from $271.21 to $276.54 million for the baseline fencing cost of $75/ft. However, the optimal fence size at $75/ft is 2,869 acres, less than 30% of the total recharge capture area. Even when fence costs are extremely low ($25/ft), the optimal fence size only slightly increases to 3,135 acres, which suggests that fully fencing the study site is never optimal over the range of parameter values we considered. As the per-unit cost of fencing is allowed to increase, the optimal fence size decreases, as does the NPV. For unit costs in excess of $287/ft, it is never optimal to build a fence of any size, as doing so would result in an NPV less than that which would be obtained by not building a fence at all (Table 2).

Table 2. Optimal fence size and net present value over a range of fence costs for the baseline 1.8% minimum growth constraint on indicator species *Gracilaria coronopifolia*

The optimal paths of groundwater extraction are illustrated in Fig. 3 for a range of fencing costs, assuming our baseline 1.8% daily growth constraint for our ecological indicator species. Optimal extraction is higher when fence costs are lower because lower costs incentivize construction of larger fences, which allow for the capture of more recharge and consequently reduce the shadow price of groundwater. Note that implementing the backstop is never optimal for the baseline growth constraint but may become optimal for stricter constraints under certain conditions. As we discuss later, sensitivity analysis shows that generally, tighter ecological constraints require more fencing in the optimal solution and are more likely to result in supplementation of groundwater extraction with the backstop resource.

Figure 3. Groundwater extraction for a range of fencing costs, assuming a 1.8% minimum growth constraint on indicator species *Gracilaria coronopifolia*.

Groundwater head levels respond as expected to changes in extraction and algae growth constraints. When no constraint is imposed, the optimal head level declines to 4.6 ft by year 50. Enforcing the 1.8% baseline algae growth constraint requires the head level to not fall below 4.97 ft, thus reducing the amount available for extraction over 50 years. Investing in fencing while enforcing the constraint, however, allows for higher water use while maintaining the head level at or above 4.97 ft.

Sensitivity Analysis

Given the many components of the linked resource-ecological-economic system, it is useful to understand how the optimal solution responds to changes in key parameter values. The purpose of this section is to broadly examine the sensitivity of our results to assumptions about each of the sub-models comprising the mountain-to-sea system: the upstream component (watershed), the central component (coastal aquifer), and the downstream component (nearshore ecosystem). We present detailed results for three key parameters that are each representative of one of the three system components: the recharge function, water demand growth, and the ecological constraint. Additional results for the height-to-volume aquifer conversion factor, aquifer leakage function, and nearshore salinity function are presented in the appendix.

Sensitivity to the ecological constraint

We start by looking at how optimal fence installation responds to tightening or relaxing of the ecological constraint. Recall that for our baseline case with a 1.8% growth constraint, fully fencing the recharge capture area is never optimal, even when the unit cost of fencing is very low. On the other hand, if the per-unit fence costs exceed \$287/ft, no amount of fencing is optimal; NPV is maximized by not building a fence. Reducing the minimum required growth rate by 5% of the baseline rate (from 1.8% to 1.7%), that is, allowing for slower algae growth due to less freshwater leakage and therefore higher coastal salinity levels, makes watershed investment undesirable, even when fence costs are very low (<\$25/ft). When we tighten the ecological constraint, the optimal fence size increases over the entire range of fence costs. Maintaining faster algae growth (≥1.9%) requires supplementing groundwater extraction with costly desalination, which means that the potential benefit of partially offsetting those costs via fencing is substantial. Therefore, within the range of fence costs that we believe is most plausible for the region (\$49-82/ft), fencing between 60-100% of the recharge zone is optimal for algae growth constraints between 1.9-2.0%. Fig. 4a illustrates how the optimal fence size changes as fence costs are varied, while maintaining different ecological growth constraints.

Figure 4. (a) Optimal fence size and (b) optimal water use for different growth constraints on ecological indicator species

When the minimum required algae growth rate is reduced by 5% of the baseline rate (from 1.8% to 1.7%), the optimal solution requires no fence installation for the baseline fencing cost of $75/ft (Fig. 4a). In that case, optimal water use declines smoothly toward its optimal steady state rate (Fig. 4b). If algae growth needs to be maintained at a higher level, fencing becomes optimal and water use declines at a slower rate. Even with a stricter ecological constraint, building a fence allows for higher water use over time in the optimal solution (Fig. 4b) because the additional captured recharge partially offsets the reduction in extraction that would otherwise be required to satisfy the growth constraint.

Sensitivity to the recharge function

Although the baseline recharge function is constructed using regional watershed management and hydrological data, i.e. it is grounded in real world data, substantial uncertainty remains regarding the extent to which watershed conservation activities like installation of ungulate-proof fencing enhances groundwater recharge. With that in mind, we rerun the optimization simulation for two additional recharge scenarios: (i) the range of recharge quantities obtained from investment in fencing is 50% higher than the baseline case (0.0024-0.3424 MG/acre/yr), and (ii) the range of recharge quantities is 50% lower (0.0006-0.0856 MG/acre/yr). Assuming again that the marginal recharge quantity gained decreases linearly with fence stock, the corresponding recharge functions are $R^{high} = 3992.7 + 0.258N_t - 0.000012N_t^2$ and $R^{low} = 3992.7 + 0.086N_t - 0.000004N_t^2$.

When the amount of recharge obtained from watershed conservation is high, the optimal fence size is lower than the baseline case by roughly 1,000 acres over a wide range of fence costs (Fig. 5a). More bang for your buck means that less fencing is needed to sustain similar levels of water use. When the amount of recharge obtained from management is low, however, the optimal fence size is always higher than the baseline case, although the difference in size varies widely over the range of fence costs considered (Fig. 5a). Optimal water use is higher when captured recharge is higher, even though the optimal fence size is smaller (Fig. 5b).

Figure 5. (a) Optimal fence size and (b) optimal water use for different recharge functions

Sensitivity to growth in demand for water

Urban development is expected to expand in the future at our study site, although the rate of expansion and timing of construction remain largely uncertain. To consider different possible consumption futures, we compare results for annual water demand growth rates of $\alpha = 0$ (baseline), $\alpha = 0.25\%$ (low), $\alpha = 0.5\%$ (moderate), and $\alpha = 1\%$ (high).[3] For the baseline algae growth constraint (1.8% growth), high water demand growth incentivizes fencing almost the entire recharge area at the baseline fence cost of \$75/ft; when fence costs are $\leq$ \$50/ft, the optimal solution entails fully fencing the recharge area. In the low and moderate demand growth scenarios, the optimal fence size is positive but always less than the full

[3] If water demand growth is higher than 1%, fully fencing the recharge area is always expected to be the optimal solution, even when the cost of fencing is high.

recharge area. In general, the optimal fence size increases with the rate of demand growth (Fig. 6a). Optimal groundwater extraction and use may increase over time if demand growth is positive because more water is consumed in aggregate at any given price point in the future when demand is higher (Fig. 6b). In addition, the entire extraction path tends to shift upward as the growth rate of demand is increased. Although higher demand in earlier periods has the potential to drive up resource scarcity, the effect is mitigated by heavy investment in fencing, ultimately allowing for sustained rates of extraction in excess of the baseline. Desalination is never optimally used for all demand growth rates considered when the other model parameters are maintained at their baseline values.

Figure 6. (a) Optimal fence size and (b) optimal water use for different rates of water demand growth

Discussion

Results from our illustrative numerical simulation suggest that for a plausible range of fence installation and maintenance costs,[4] partial fencing is always optimal, i.e., fully fencing the recharge capture area or not building a fence at all are suboptimal strategies. Sensitivity

[4] Recall that, to simplify our model and simulation, we assume that when fence is installed, routine maintenance costs are obligated. In this way, we can combine installation and maintenance costs of fencing into a present value unit cost. In a more general formulation, wherein investment can be freely applied toward new fence construction or maintenance of existing fence, the optimal fence stock may be increasing or decreasing over time. Although we do not expect our general conclusions to change in this case, we leave it to further research.

analysis also shows that the optimal fence size tends to increase as the ecological constraint becomes stricter, the optimal fence size is smaller if watershed management is more efficient at capturing recharge, and higher demand growth requires a larger fence in the optimal solution. In every scenario, fence costs and optimal fence size are inversely related, but the cost effect tends to be small relative to the effect of the other three factors. This is perhaps most clearly illustrated in Fig. 4a, 5a, and 6a. Within what we believe to be the most plausible range of fencing costs ($49-82/ft), the optimal fence curves are fairly flat, while the vertical distance between curves tends to be large. Thus, an integrated management approach would likely benefit more from allocating effort toward reducing uncertainty surrounding ecological linkages, recharge capture, and demand growth than it would from trying to improve the accuracy of fencing cost projections.

Although the optimal solution for our baseline scenario is to invest in fencing to enclose approximately 3,000 acres or 30% of the total recharge capture area, our sensitivity analysis shows that slight perturbations of the baseline parameter values can shift the optimum toward full fencing. For example, adjusting demand growth from zero to 1% results in such a shift, assuming a unit fencing cost of $75/ft. From a management standpoint, taking a precautionary approach and leaning toward full fencing may be desirable, depending on the degree of confidence the manager has in his or her knowledge of the key parameter values in the system. One challenge with this approach, however, is the potential opportunity cost of alternative land uses. Many of the high-elevation forested areas in Hawaiʻi are already zoned as protected watershed areas, so land use tradeoffs are less relevant in our application. But more generally, forest protection efforts often end up in competition with agriculture and urban land uses. In that case, the NPV formulation should be adjusted to account for the opportunity cost of the land being considered for watershed protection activities, and all else equal, the optimal fence size would likely be smaller.

Because the keystone ecological species at our study site is primarily valued for its cultural and traditional uses, directly quantifying the monetary value of protecting the species is challenging, given the data available for our analysis. Our results do indicate, however, that maintaining the baseline ecological growth constraint (without watershed conservation) reduces NPV by \$11.57 million, which illustrates the tradeoff between household water consumption and the growth rate of algae. Increasing the minimum required algae growth rate by 10% of the baseline rate (from 1.8% to 2.0%) magnifies that loss to \$96.74 million. Therefore, if water users are willing to forego the amount of groundwater extraction required to protect the ecological species, we can say that they collectively value the algae at no less than \$12-97 million. When the simulation allows for watershed conservation, optimal investment is positive and offsets approximately \$8.21 million that would be forgone under baseline conditions upon imposition of the ecological constraint. Since optimal investment increases total NPV relative to the baseline, water users in aggregate benefit from such activities, and a payment for ecosystem services (PES) program could serve as a mechanism to finance watershed investment through water user payments. In general, PES programs incur transaction and compliance costs, in addition to opportunity costs for service providers, all of which should be accounted for in program design. At our particular study site, however, the recharge capture areas under consideration are primarily already zoned for conservation. Thus, perhaps the bigger challenge in our application is how best to reconcile the mismatch of fence costs and water benefits over time. To ensure that earlier generations of water users are not unfairly bearing a disproportionate share of the financial burden, a dynamic PES approach may be warranted (Roumasset and Wada, 2013), wherein bond-financing allows for the assignment of user fees in proportion to benefits received over time.

This study has some potential limitations, particularly with regard to assumptions about the functional forms and associated parameters of the various equations describing our mountain-to-sea system. Linear and quadratic forms simplify the solution to the optimal control problem and are amenable to relatively straightforward sensitivity analysis. However, abstracting from higher-order nonlinearities may have implications for optimal trajectories. For example, if the conservation capital stock equation or investment cost function were not linear in investment, the optimal fence solution would no longer be described by a most rapid approach path. Moreover, limiting the flexibility of functional forms and corresponding parameters reduces the generalizability of our results to locations outside of our study site. Nevertheless, we feel that the developed mountain-to-sea methodology is transferable to other areas where resource managers are interested in coordinating the stewardship of multiple resources, such that improvement of an upstream resource confers benefits to a downstream resource via a mutually linked intermediate resource.

Conclusion

To improve our understanding of the tradeoffs involved in dynamic mountain-to-sea management, we integrated hydrologic ecosystem service provision via forested watershed protection, maintenance of a nearshore marine GDE, and coastal groundwater withdrawals into a joint dynamic optimization framework. Our results suggest that investment in watershed protection can be effective at mitigating the cost imposed on groundwater users of maintaining the linked nearshore ecosystem. We show that after taking into account the cost of optimally investing in watershed conservation, groundwater users are at least as well off in aggregate after the GDE constraint is imposed as they are if no investment is made. That is, NPV is higher with optimal investment. However, financing the optimal joint management strategy may require some form of borrowing (e.g., bond-financing), given that investment costs are primarily incurred in earlier periods, while benefits to water users of reduced groundwater scarcity occur largely in the future.

Author Contribution Statement

Christopher Wada: Conceptualization, Methodology, Formal Analysis, Writing – Original Draft, Writing – Review & Editing.

SIttidaj Pongkijvorasin: Conceptualization, Methodology, Formal Analysis, Writing – Original Draft, Writing – Review & Editing, Visualization, Funding Acquisition.

Kimberly Burnett: Conceptualization, Writing – Review & Editing, Supervision, Project Administration, Funding Acquisition.

Declaration of Interests

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix

Given the many assumptions underlying our numerical model, it is important to understand how robust the baseline results are to variations in key parameter values. We present results for sensitivity analyses of the height-to-volume aquifer conversion parameter, the aquifer coastal leakage parameter, and the nearshore salinity coefficient. Overall, the results suggest that for reasonably small variations in each of the parameter values ($\leq$ +/- 25% of the baseline value), the resulting NPV is not substantially lower than the baseline value ($279.42 million), although in some cases the optimal solution requires heavier investment in fencing to avoid a decline in NPV. Results for all cases are summarized in Table 3.

Table 3. Sensitivity analysis: net present value, optimal fence size, and benefit of fencing over a range of values for the aquifer height-to-volume conversion, aquifer leakage, and nearshore salinity parameters.

Given the relatively calm sea conditions, homogeneous coastline geology, lack of surface water inputs or outputs, and unidirectional flow to the ocean along the coastline at our study site, we believe that the sharp-interface assumption is sufficient for describing average changes in water volume and head level in our model. Nevertheless, given that the interface between fresh and saltwater is not completely sharp in reality, we rerun the simulation assuming that the head-to-volume conversion parameter is +/- 25% of its baseline value ($\gamma = 0.000049$). As expected, when the value of γ is 25% higher, i.e. when changes in groundwater volume have a larger effect on head level, less water is available, NPV without investment in fencing ($268.75

million) is lower, and optimal investment in fencing (2967 acres) is higher than in the baseline case. When the value of γ is 25% lower, more water is available, NPV without investment in fencing is higher ($273.48 million), and optimal investment in fencing (2689 acres) is lower than in the baseline case. The benefit of fencing as a percentage of NPV is fairly robust to perturbations in the value of γ, falling in the range of 2-4%, compared to the baseline value of 3%.

Next we test the sensitivity of our results to variations in the aquifer leakage coefficient. Recall that leakage from the aquifer along the coast is assumed to be a function of the head level: $l(h_t) = zh_t^2$, where $z = 117.8$ in the baseline scenario. Pongkijvorasin et al. (2010) estimate that the value of the parameter z is uncertain and most likely falls in the range of 113 to 121. We consider an even wider range in our sensitivity analysis, by rerunning the simulation using values of z +/- 25% of the baseline level. When the value of z is 25% higher, i.e. more leakage occurs at any given head level, it is easier to meet the ecological constraint, NPV without investment in fencing is higher ($273.47 million), and optimal investment in fencing (2820 acres) is lower than in the baseline case. When the value of z is 25% lower, it is more difficult to meet the ecological constraint, NPV without investment in fencing is lower ($256.28 million), and optimal investment in fencing (3254 acres) is higher than in the baseline case. The benefit of fencing as a percentage of NPV is fairly robust to perturbations in the value of z, ranging from 2-8%, compared to the baseline value of 3%. Note, however, that the benefit of fencing is disproportionately larger when the leakage coefficient is lower because the challenge of meeting the ecological constraint drives up the scarcity value of water more rapidly in that case.

Lastly, we examine the robustness of our results to different values of the salinity coefficient. Recall that nearshore salinity is modeled as a linear function of aquifer leakage at the coast: $s_t = 36 - 0.00125l(h_t)$. We find that even for a relatively small perturbation (+/-10%) to the baseline value of 0.00125, the effect on outcomes is quite substantial. When the salinity coefficient is 10% higher, i.e. nearshore salinity is lower for a given rate of leakage, it is easier to meet the ecological constraint, NPV without investment in fencing is higher ($281.72 million), and no fencing is required in the optimal solution. When the salinity coefficient is 10% lower, i.e. nearshore salinity is higher at any rate of leakage, NPV without investment in fencing is lower ($235.6 million), and optimal investment in fencing (6297 acres) is substantially higher than in the baseline case. The benefit of fencing as a percentage of NPV is almost 18% in the low salinity coefficient case, nearly six times the baseline value of 3%, which suggests that our results are particularly sensitive to the salinity assumption.

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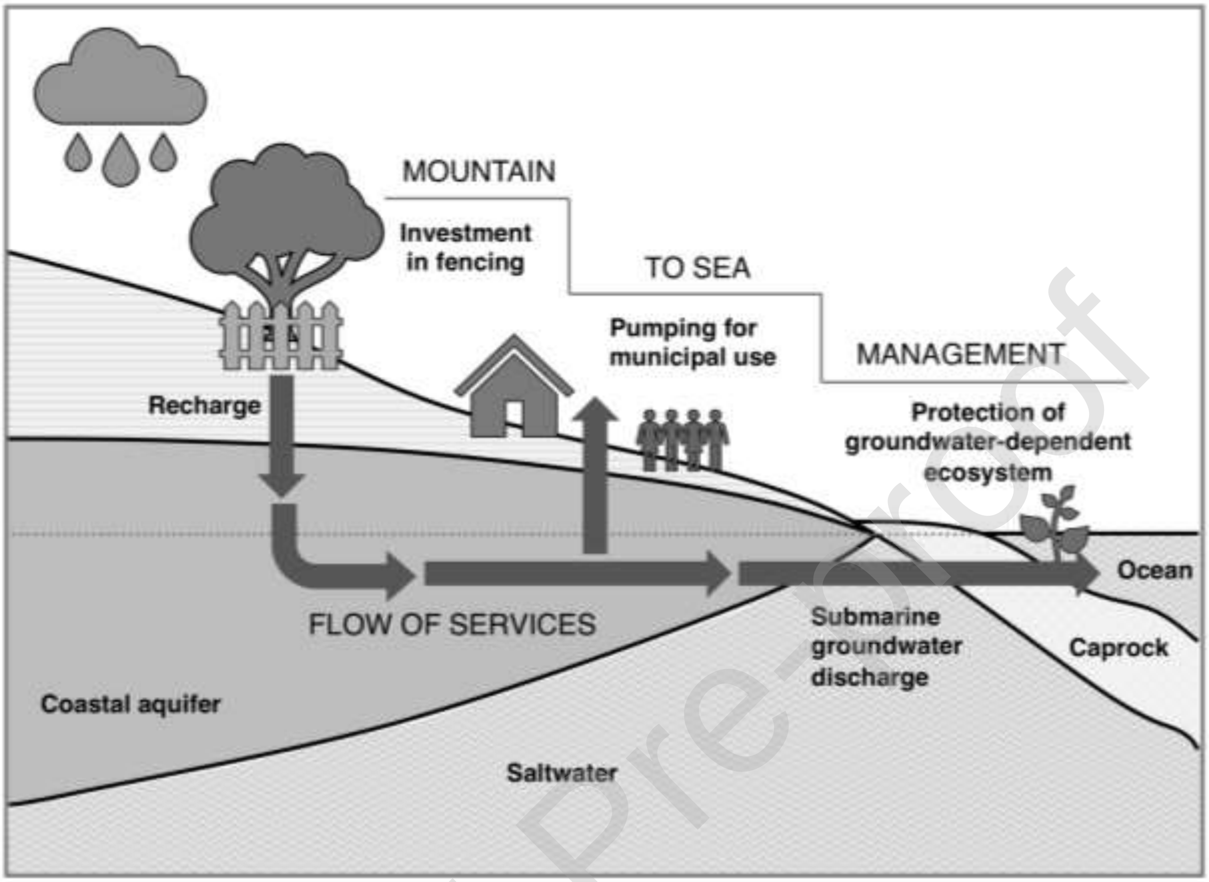


Figure 1. Mountain to sea management of a forested watershed, coastal aquifer, and groundwater-dependent ecosystem.

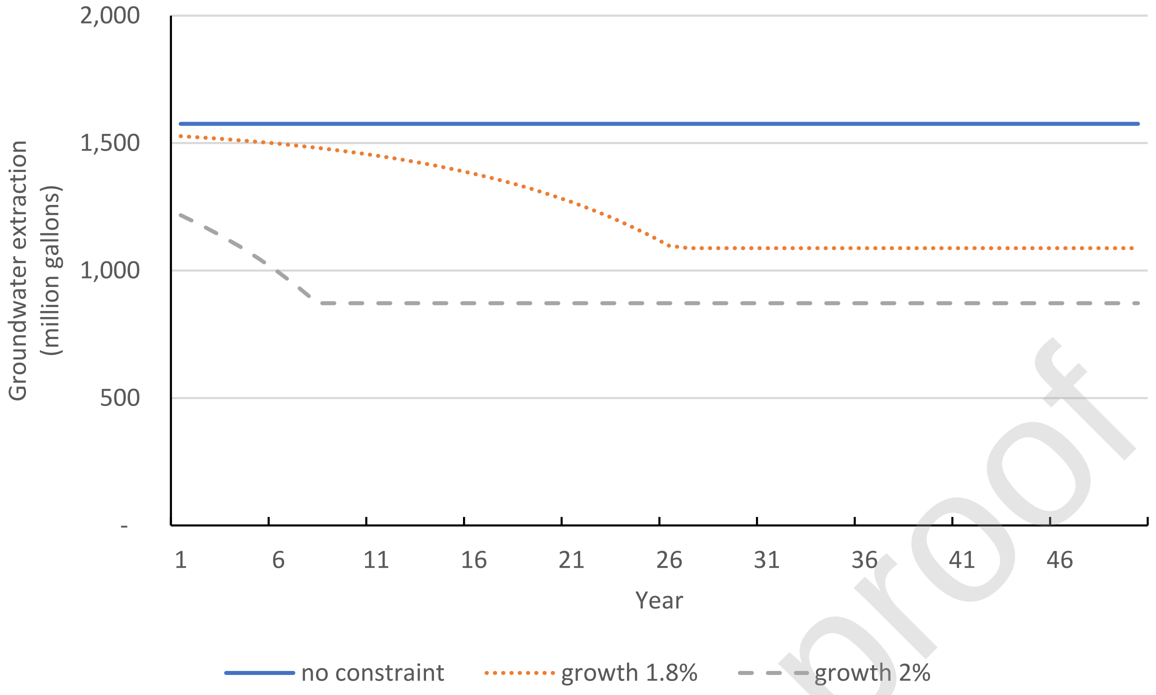


Figure 2. Groundwater extraction for the unconstrained optimum and two growth constraints on indicator species *Gracilaria coronopifolia* with no fencing.

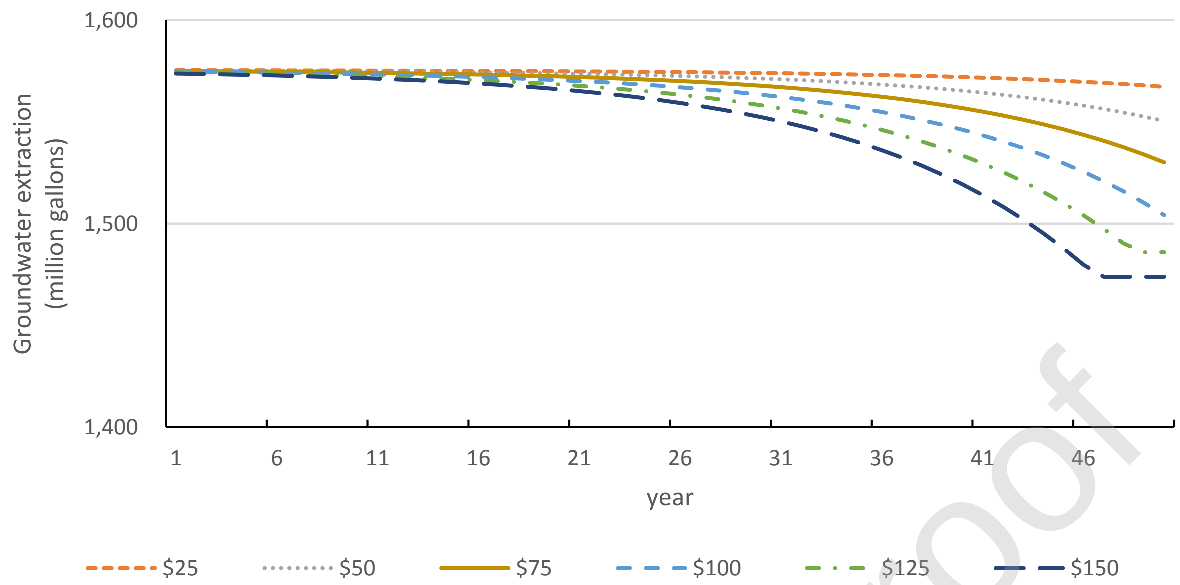
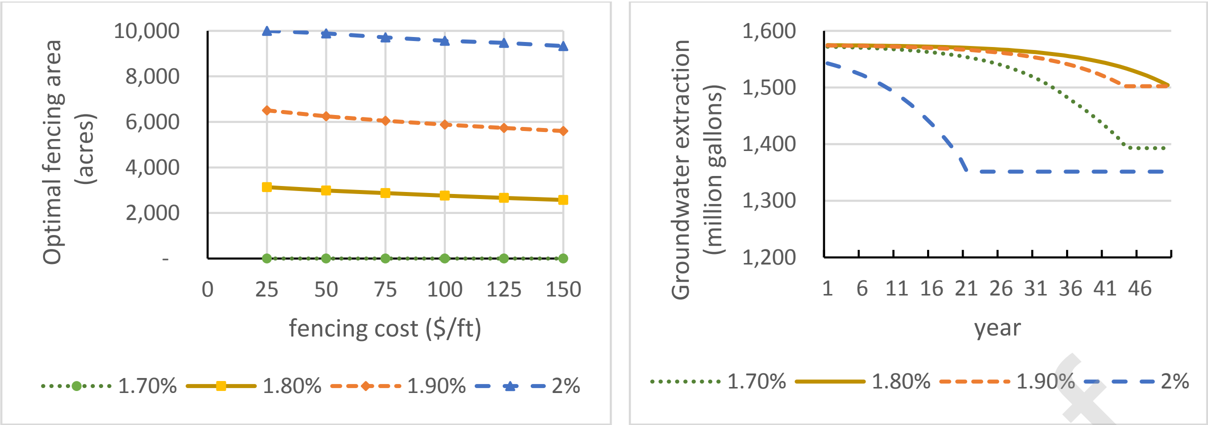
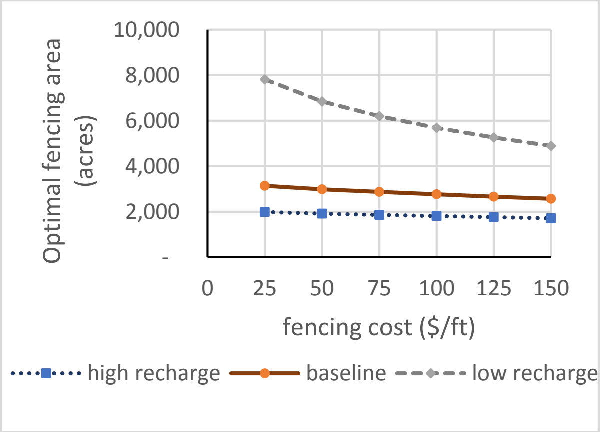


Figure 3. Groundwater extraction for a range of fencing costs, assuming a 1.8% minimum growth constraint on indicator species *Gracilaria coronopifolia*.

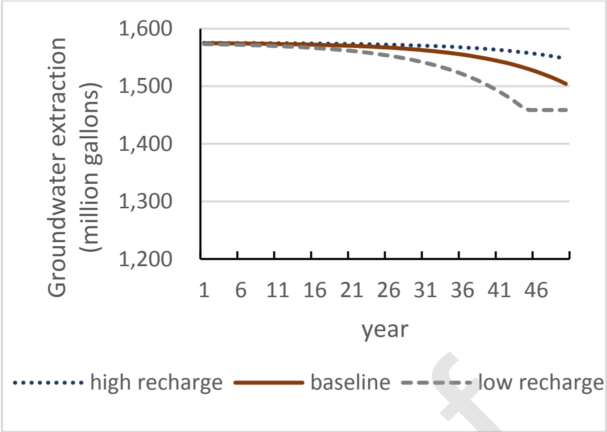


(a) (b)

Figure 4. (a) Optimal fence size and (b) optimal water use for different growth constraints on ecological indicator species



(b)



(e)

Figure 5. (a) Optimal fence size and (b) optimal water use for different recharge functions

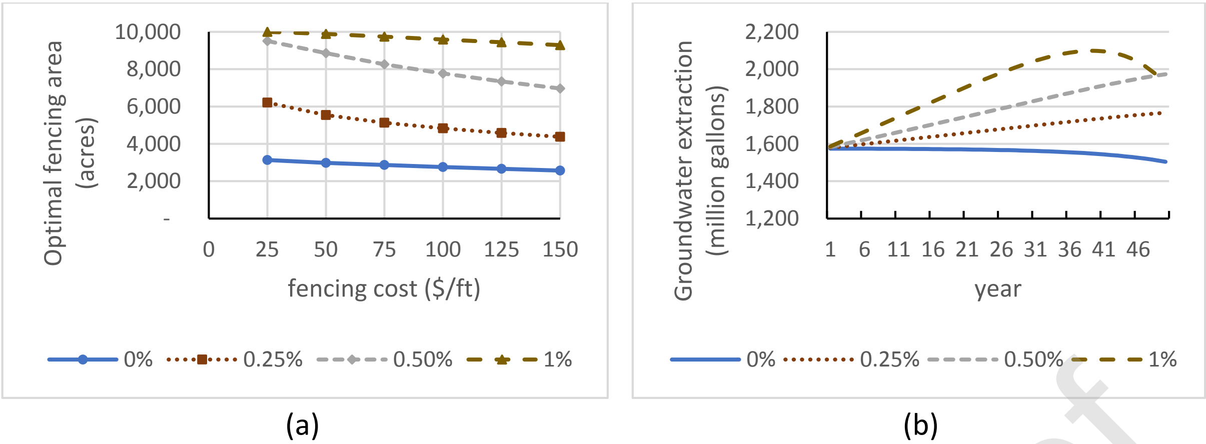


Figure 6. (a) Optimal fence size and (b) optimal water use for different rates of water demand growth

Description	Equation
Aquifer state equation	$\dot{h}_t = (0.000049)(R(N_t) - l(h_t) - q_t)$
Recharge	$R(N_t) = 3992.7 + 0.172N_t - 0.000008N_t^2$
Coastal discharge	$l(h_t) = 117.8h_t^2$
Groundwater unit extraction cost	$c(h_t) = 0.00115(1322.82 - h_t)$
Water demand	$p_t = 15.066 - 0.0086q_t e^{-\alpha t}$
Nearshore salinity level	$s_t = 36 - 0.00125l(h_t)$
Growth rate of algae	$g_t = 10.2975 - 0.2625s_t$

Table 1. Key equations summarizing the mountain-to-sea system

Per-unit cost ($/ft/50-yr)	First-year installation size (acres)	NPV over 50 years (million $)	Increase in NPV compared to no fence (million $, %)
25	3,135	281.62	10.41 (3.8%)
50	2,984	280.51	9.30 (3.4%)
75	**2,869**	**279.42**	8.21 (3.0%)
100	2,761	278.37	7.16 (2.6%)
125	2,661	277.34	6.13 (2.3%)
150	2,567	276.33	5.12 (1.9%)
287	2,101	271.23	0.02 (0.0%)
≥288	0	271.21	0 (0.0%)
Benchmark no fence	**0**	**271.21**	-

Table 2. Optimal fence size and net present value over a range of fence costs for the baseline 1.8% minimum growth constraint on indicator species *Gracilaria coronopifolia*

Parameter	Relative to baseline	NPV without fence (million $)	NPV with optimal fence (million $)	Optimal fence installation size (acres)	Benefit of fencing (million $, %)
Aquifer height-to-volume	-25%	273.48	279.54	2,689.35	6.06 (2.2%)
	25%	268.75	279.35	2,967.41	10.60 (4.0%)
Leakage	-25%	256.28	278.72	3,253.86	22.44 (8.8%)
	25%	273.47	279.44	2,820.15	5.96 (2.2%)
Salinity	-10%	235.60	277.78	6,296.71	42.18 (17.9%)
	10%	281.72	281.72	-	-
Baseline		***271.21***	***279.42***	***2,869.08***	***8.21 (3.0%)***

Table 3. Sensitivity analysis: net present value, optimal fence size, and benefit of fencing over a range of values for the aquifer height-to-volume conversion, aquifer leakage, and nearshore salinity parameters.