Modeling the Wetland Mitigation Process: A New Dynamic Vision of No Net Loss Policy

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Abstract

Over the last two hundred years, United States has experienced dramatic wetland losses in terms of both quality and extent. In 1987, the National Wetlands Policy Forum recommended that U.S. wetlands policy should achieve overall "no net loss" of the country's remaining wetland acreage and function. Since then, regulations requiring compensatory mitigation for wetland losses, often through wetland creation or restoration, have become an essential component of federal wetland protection efforts. Recent reports have concluded that no net loss policy has been successful, citing the virtual elimination of wetland losses experienced in certain areas. However, these reports have not assessed the temporal nature of wetland loss and restoration. Delays in initiating and completing restoration activities mean that frequent temporary wetland losses can contribute to a consistent net loss over time. This paper analyzes wetland loss and compensation as dynamic processes that include temporal lags endemic to various mitigation techniques. Here, a system dynamics model of the mitigation process is used to explore wetland alteration and mitigation data collected between 1993 and 2004 for the Chicago, IL region. By analyzing wetland change dynamically, it becomes possible to adjust wetland mitigation methods to more effectively eliminate temporal net loss of wetlands.

Keywords: Wetland Mitigation, No Net Loss, System Dynamics, Land Use and Environmental Planning, Ecological Restoration

Introduction

Urban and agricultural development has had widespread and irreversible impacts on the extent and quality of wetlands around the world (Baldock 1984; OECD 1992). In 1987, the U.S. Environmental Protection Agency (EPA) convened the National Wetlands Policy Forum (NWPF), a wide array of stakeholders whose goal was to "address major policy concerns about how the nation should protect and manage it valuable wetlands resources (National Wetlands Policy Forum 1988, pg. vii)." The forum, which was comprised of government, industry, and environmental leaders, as well as ranchers and academic experts, attempted to refocus United States wetland regulation towards a policy of "no net loss," recommending that,

> …the nation establish a national wetlands protection policy to achieve no overall net loss of the nation's remaining wetland base, as defined by acreage and function, and to restore and create wetlands, where feasible, to increase the quality and quantity of the nation's wetland resource base (National Wetlands Policy Forum 1988, pg. 3).

Since the NWPF, the policy goal of "no net loss" of wetlands has become a driving force behind wetlands management throughout the United States (Hansen 2006). The wetland mitigation permitting program established under Section 404 of the Clean Water Act and administered by the Army Corps of Engineers (Corps) has become increasingly responsible for sustaining no net loss policy (Goldman-Carter 1992; Turner et al. 2001; Tolman 2004). Under compensatory mitigation regulations, wetland losses can theoretically be offset by requiring anyone responsible for wetland destruction to create, restore, or preserve wetlands in another area.

In order to accurately assess the aggregate effects of wetland alterations, as well as the status of no net loss, any system that tracks wetland losses and gains must take into account the inherent delays in land alteration and restoration projects. Although regulatory permits view wetland destruction and compensatory mitigation as concurrent and instantaneous, delays in initiating and completing restoration activities mean that large numbers of temporary wetland losses can compound into a consistent, temporary net loss of wetland acreage and function over time.

Although significant work has addressed the ecological issues of restoration behavior at the scale of individual wetlands (Sklar et al. 1985; Costanza et al. 1990), little work has focused on the aggregate, dynamic behavior of wetland loss and gain at the landscape level. As a result, several questions remain largely unaddressed. As a steady stream of wetlands are destroyed and their restoration is initiated, under what conditions will the landscape experience a temporary net loss of wetlands? Is it possible to prevent this from occurring? If so, can preventative methods actually be put into practice as applicable, enforceable policies at the national, state, and local levels?

I address these questions through the analysis of a system dynamics model of the wetland compensatory mitigation process and its effects on temporary wetland loss. This type of investigation helps us to explicitly understand mitigation processes, as well as policies and environmental variables that affect wetland loss and gain, as they progress over time. This model includes vital factors associated with mitigation policy, including mitigation failure rates, varying mitigation ratios, and the temporal lags inherent in the wetland restoration process. Here, I analyze distinct wetland mitigation techniques while applying this model to the Chicago, IL region using wetland alteration and mitigation data collected between 1993 and 2004.

This article begins with a discussion of the history and implementation of no net loss policies, highlighting the dynamic nature of wetland destruction and creation. Next, I apply this discussion to the Chicago region, focusing on the network of wetland mitigation regulations that have been established to compensate for new wetland alterations. I then introduce a system dynamics model for analyzing wetland loss and restoration dynamics, testing several scenarios based on currently regulatory assumptions about restoration efficiency and success rates. Finally, I discuss the implications of these scenarios for future regulations at the federal, state, and local levels.

Background

The Role of Wetland Conservation

In the continental United States, over 53 percent of all naturally-occurring wetlands (more than 117 million acres) were converted into agricultural and urban land uses between 1780 and 1980 (Dahl 1990). However, ecological research has revealed wetlands to be extraordinarily productive ecosystems that perform a wide array of ecological functions including carbon sequestration, flood attenuation, wildlife habitat and open space provision, and water quality improvement (NRC 1992, 2001; Cylinder et al. 2004). Additionally, many studies have shown that the loss of these wetland functions can have significant repercussions on the hydrological and ecological stability of the landscape (Hulsey and Tichenor 2000; Arnold 2006).

In 1987, the NWPF recommended that national wetland policy “pay particular attention to, and explicitly evaluate, the cumulative effects of various types of alterations on the systems under study (pg. 20).” During the intervening years, one important method of protecting against cumulative losses has been the widespread establishment and implementation of compensatory mitigation regulations.

Compensatory Wetland Mitigation

In 1990, the Corps and EPA formerly endorsed no net loss, creating the first regulatory guidance document that uniformly established the wetland mitigation process as a national policy (Corps and EPA 1990). Here, developers, or anyone else altering a wetland, must avoid wetland impacts to the maximum extent practicable. Developers must then take steps to minimize unavoidable impacts, and finally, if necessary, provide compensation for unavoidable wetland impacts. Regulators can then grant a permit allowing developers to alter or destroy wetlands on the condition that compensatory mitigation is performed, usually in the form of wetland restoration or creation.[1] Here, restoration and creation lie on a continuum of desired types of ecological function, with fully functional wetlands on one end, and completely converted wetlands on the other (Jackson 1995; Bradshaw 1996).

Several methods of mitigating wetlands have been developed since the late-1970s. These methods can be categorized as permittee-responsible mitigation (PRM) and third party mitigation.

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[1] There is always some uncertainty that mitigation projects may fail to reach their stated ecological goals. This uncertainty is often reduced through financial bonding requirements (Corps 1997).

Permittee-Responsible Mitigation

Under PRM, individual land developers are required to restore, create, or preserve alternate wetlands either on the same development site, or at another suitable location. Here, wetland alterations and the start of mitigation activities are generally understood to occur simultaneously.

Third Party Mitigation Methods

Third party mitigation methods were originally devised to improve the likelihood of mitigation success. In recent years, support for third party mitigation has grown among regulators and developers alike (ELI 2002; Shabman and Scodari 2004; BenDor and Brozovic 2007). These methods can be divided into 'wetland mitigation banking' and 'in-lieu fee mitigation.' Although these techniques may appear to yield similar results in terms of net acreage lost, their timelines for wetland restoration actually act in reverse of each other, creating major differences in their dynamic wetland restoration behavior.

Wetland Mitigation Banking

Wetland mitigation banking has been defined by the Corps and EPA (1995, pg. 58605) as mitigation that takes place "in advance of authorized impacts to similar resources." Commonly under mitigation banking, a third party entrepreneur ("the mitigation banker") obtains authorization from regulators to create or restore a relatively large area of wetlands. These wetlands are then used as a 'bank' of credits and are sold to developers that use them to satisfy their mitigation obligations to regulators (Bonds and Pompe 2003). Over the last fifteen years, banking has drawn increasing support from regulators, who are able to establish higher ecological standards for banks since banks provide mitigation for multiple projects (Corps and EPA 1995; Shabman et al. 1996; Scodari and Shabman 2001; Shabman and Scodari 2004; Corps 2006).

However, a closer look at bank implementation programs reveals the rarity with which banks actually complete mitigation prior to impacts. Robertson (2004, 2006) demonstrated that 60 percent of all credit sales in the ACOE Chicago District between 1994 and 2002 occurred in banks that had not even achieved their initial ecological performance standards. This behavior is probably due to difficulties that entrepreneurs have in entering the wetland banking industry. Here, high performance bonding requirements, combined with major upfront investments for land purchase and restoration, present steep barriers to market entry. As a result, the Corps has allowed banks to phase their credit sales, releasing credits before all ecological standards have been met. In Chicago, banks are allowed to sell up to 70 percent of their credits before they achieve full functional establishment (Corps 1997).

In-Lieu Fee Mitigation

In-lieu fee (ILF) programs typically function through agreements made between regulators and a public agency or non-profit organization, whose job it is it perform wetland restoration or creation (ELI 2006). Under this system, rather than performing their own wetland restoration, developers issue a cash payment to ILF programs in order to satisfy their mitigation requirements (ELI 2002). ILF programs usually lack an initial endowment and frequently rely on fee revenues to provide funds for compensation activities (Urban et al. 1999). As a result, ILF program sponsors typically pool funds from multiple developers to gain enough capital to purchase mitigation sites and begin restoration activities.

The time taken to pool funds usually creates an additional time lag between permitted wetland fills and implemented compensation actions, particularly in the initial years of the ILF program. As a result, ILF programs end up beginning mitigation activities at some point after development activities begin, thereby exacerbating the same temporary net functional losses as seen with permittee responsible mitigation (ELI 2002). This contrasts with mitigation banking in that banking requires an initial investment, with at least some mitigation taking place before it can be used to offset wetland impacts.

Restoration Dynamics

During wetland creation or restoration, there is commonly a long period during which wetland hydrology, soils, and vegetative communities establish/re-establish themselves (Richardson 1994; Gutrich and Hitzhusen 2004; Klimas 2004). These temporal lags slow the reestablishment of wetland functions. In the case of non-bank mitigation, this period occurs after construction has taken place and the original wetland is altered. Therefore, any loss in wetland services over time is a function of the point at which one wetland is destroyed and the time taken for the mitigation wetland to attain full function. Both of these factors, as well as the inherent uncertainty of ecological restoration outcomes, influence the temporary net loss of function that occurs in the wake of mitigation projects.

The NWPF acknowledged this issue and its effects on no net loss, stating a preference that, "…to the extent feasible, any required compensation be under-taken before the permitted wetlands alterations occur (National Wetlands Policy Forum 1988, pg. 44)." As a result, regulators now require a larger amount wetland area than previously existed in order to partially account for the temporal loss between wetland impact and wetland compensation (Corps and EPA 1990)[1]. This increase in required area is known as a 'mitigation ratio,' and is defined as the ratio of mitigated to altered wetland area.

Research on the effects of mitigation on temporal net loss remains relatively sparse. In one recent study, Gutrich and Hitzhusen (2004) used case studies of two wetland complexes to understand the monetary cost of time lags associated with wetland loss and re-establishment. Here, the authors found that it required a median of 33 years and 13 years for floral and soil ecosystems to achieve full functional equivalency under logarithmic growth models in Ohio and Colorado, respectively. By using results from prior wetland valuation studies, they also estimated the average economic costs from restoration lags in Ohio and Colorado at \$16,640 and \$27,392 (2000 US\$), which are equivalent to 25% and 49% of the total restoration costs, respectively. These results suggest that, due to the application of no net loss policy and mitigation regulations, society bears significant costs associated with lost wetland benefits due to the time lags inherent to mitigation site restoration projects.

Finally, the explorations of restoration dynamics by Klimas (2004), Klimas et al. (2004, 2005), and Richardson (1994) also contribute to this topic by defining a 'functional trajectories' concept[2]. Here, functional trajectories describe paths taken by restored wetland functions as they gradually grow to offset the functional losses of altered wetlands (Aronson and Le Floc'h 1996; Bradshaw 1996; Hobbs and Harris 2001). The functional trajectories concept has been challenged repeatedly in the literature on restoration ecology based on its reliance on outdated Clementsian ecology (Clements 1916; Gleason 1917) which views restoration as orderly,

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[2] These paths are similar to paths of ecological succession, where succession is generally thought of as the "natural process, following a disturbance, by which one community of plants and animals gradually replaces another, in response to changing environmental conditions (Helms 1998)."

predictable, and deterministic (McIntosh 1980; Zedler and Callaway 1999). However, in dealing with aggregate interpretations of wetland destruction and restoration, regulators view restoration as an orderly, attainable process with a stated, deterministic goal (Corps and EPA 1990, 1995). As a result, the literature on functional trajectories is useful in that it directly correlates with currently regulatory involvement in the wetland mitigation process.

Modeling the Dynamics of Wetland Mitigation

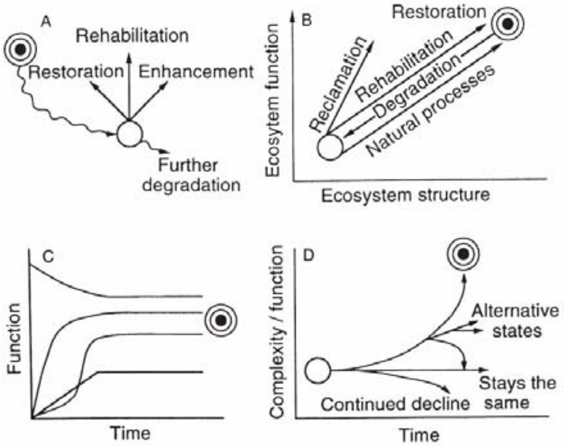
A Wetland Impact and Mitigation Response Model

A major assumption of any model of the mitigation process involves the dynamic behavior of wetland restoration as a whole. Here, wetland restoration comprises the growth of a number of different functions (including hydrologic functions, soil microbiology, floral richness, etc.), each of which has its own behavior.

Zedler and Callaway (1999) noted that many trajectory studies assumed hypothetical models of restoration site trajectories, particularly the dynamic behavior of wetlands over time. Several of these hypothetical models are shown in

Figure 1. Several trends appear in this literature; studies by Klimas (2004), Klimas et al. (2004, 2005), and Richardson (1994) all assumed a pattern of logistic growth of wetland function. Alternatively, previous system dynamics models developed by Saeed (2004) and Gutrich and Hitzhusen (2004) have both used logarithmic growth functions. Additionally, the exact level of functional disaggregation that is necessary for estimating restoration progress for policy purposes continues to be a major avenue for further research.

Figure 1: Hypothetical Restoration Trajectory Models



Hypothetical Models of Restoration site trajectories, with natural ecosystem conditions indicated by a bull's-eye and the degraded system as an open circle. Redrawn from Magnuson et al. (1980), this figure was developed to characterize lake degradation (A). Redrawn from Bradshaw (1984) and Dobson et al. (1997), this model characterized degradation due to mining or other operations; the authors acknowledged that assistance would be needed for rapid ecosystem development (B). Redrawn from Kentula et al. (1993); the authors indicate that some attributes of constructed wetlands may initially be higher than reference systems, giving the example of Simpson's diversity index for vegetation (C). Redrawn from Hobbs and Mooney (1993) (D).

Source: Zedler and Callaway 1999, pg. 70 (Figure and Caption - will need permissions)

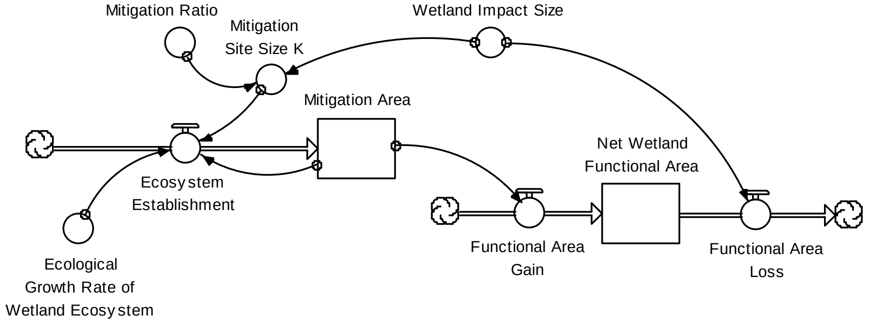
I take a highly aggregated approach to estimating restoration behavior and assume acreage to be a proxy for wetland function. Although this assumption is crude and may be inaccurate in many instances, it aligns with the assumptions of other mitigation literature as well as the assumptions and treatment of regulators throughout the United States (Salzman and Ruhl 2000, 2004). In many instances, small wetlands can attain higher levels of certain functions than much larger wetlands. Therefore, the correlation between wetland size and function is not always exact (Palmer et al. 1997; White and Walker 1997).

I also assume that wetland functions during restoration grow in a logistic fashion. This is a common assumption in the natural resource economics literature when addressing the ecosystem or population growth, as well as the exploitation of biological resources (Plourde 1970; Clark 1976; Fisher 1981; Dasgupta and Heal. 1993). Regulators often presuppose that higher mitigation ratios, which yield larger mitigation sites, will eventually create higher levels of wetland function. Here, if we assume a logistic growth behavior for wetland functional area, we would expect that the restored wetland will eventually grow to equal the functional area of the original wetland, thereby offsetting the impact. Since the 1.5:1 ratio will eventually create a larger wetland, I also assume that functional growth will take place faster than the 1:1 ratio wetland. This is a basic assumption of the logistic growth equation, where growth of an area is based on the size of the area itself.

System Dynamics

I simulate the mitigation process using the system dynamics modeling methodology (Figure 2). System dynamics uses stock-flow-feedback structures to describe and understand non-linear complex systems (Forrester 1969; Ford 1999; Sterman 2000). Here, stocks (boxes) represent accumulations of material or information (wetlands) and flows (double lines with valves) represent change in those accumulations (e.g. functional gain and loss). Flows are described by converters (circles), and generate feedback within the system through information links, represented by arrows

Figure 2: Logistic Wetland Growth Model



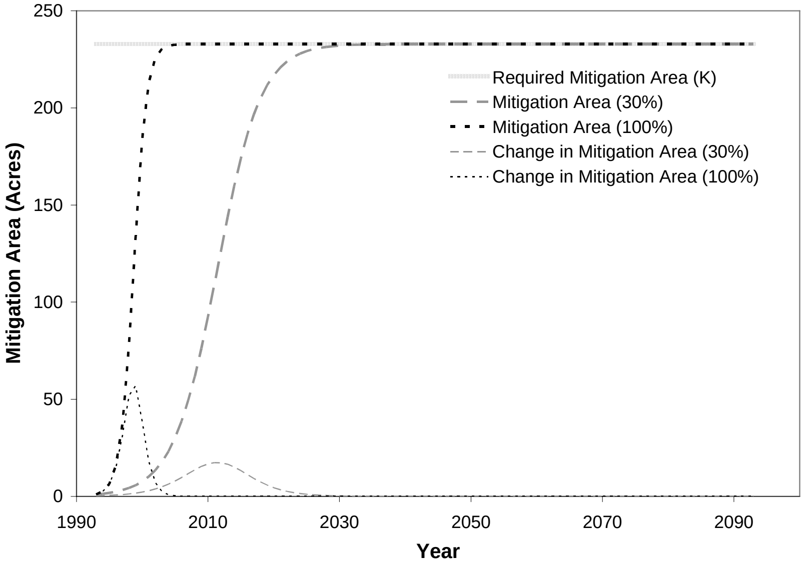
An initial system dynamics model representing this behavior is shown in Figure 2. We can observe how this framework simulates developer input into the restoration process. Given different

The heavily influenced by time, energy, and monetary investment on the part of the developer or developer's wetland consultant. The growth of the mitigation area is given by the classic logistic growth function given in Equation 1:

$$\Delta M = rM\left(1 - \frac{M}{K}\right)$$ **Equation 1**

Where: *M* = established mitigation area, *K* = mitigation site size goal that is sought by *M*, and *r* = ecological growth rate of the wetland ecosystem

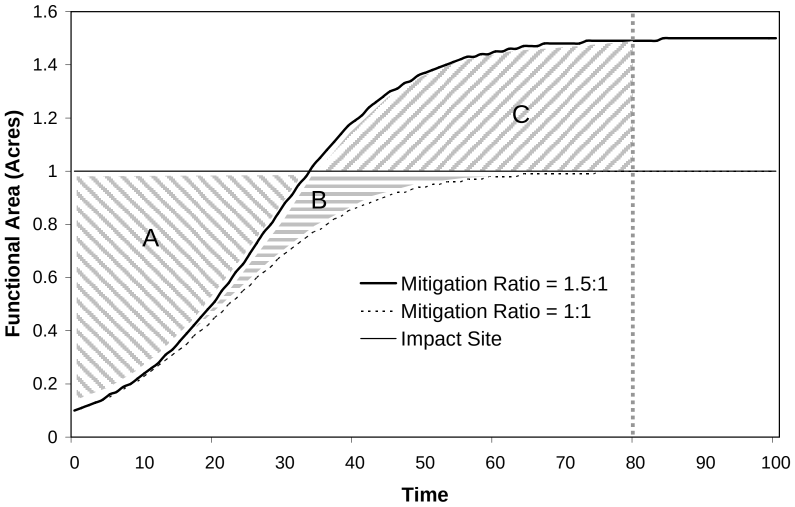
Figure 3: Logistic Pattern for Mitigation Area Growth



Notes: Tests with r set at 30% and 100%

Using this model, we can understand how create a method for comparing the dynamic behavior of wetland reestablishment where a single wetland impact is mitigated at 1:1 and 1.5:1 ratios. Under this scenario, a wetland is destroyed at time $t = 0$ and restoration begins immediately on another wetland whose initial function is equivalent to 10% of the altered wetland (Figure 4).

Figure 4: Functional Trajectories of Impact and Mitigation Wetlands



A: Loss in wetland functional area during initial years after wetland impact and mitigation when mitigation ratio is 1.5:1.

A + *B*: Loss in wetland functional area during initial years after wetland impact and mitigation when mitigation ratio is 1:1.
C: Functional area gains due to larger wetland functional area at the mitigation site (1.5:1 ratio). This holds under the assumption that the larger mitigation wetland has a larger functional capacity than the impact site (horizontal line).
t = 80: Time at which functional area gain at mitigation wetland should offset loss from impacted wetland under 1.5:1 ratio (*A* + *B* = *C*).

Area *A* denotes the initial functional area lost due to wetland alterations, taking into account the functional trajectory of mitigation under a ratio of 1:5:1. Area *A* + *B* denotes the functional area lost over time when mitigating at a ratio of 1:1. Area *C* represents the area that a ratio of 1.5:1 attains after growing above the original function of the impacted wetland. Area *C* grows as the mitigation site attains a high level of function after years of growth, eventually offsetting the initial functional loss (A) during time *t = 80.* In the case where the required ratio is 1.5:1, the higher functional capacity of the mitigated wetland means that, at some point in the future, there will eventually be a *net gain* in wetland function over time due to mitigation. Here, I again use acreage as a crude proxy for function as it has been used in the past for regulatory matters.

However, in the case where the mitigation ratio is 1:1, no gain occurs. Since the mitigated wetland's function never exceeds that of the original wetland, there is no way to offset the losses experienced after the initial wetland alteration. As a result, *A*+*B* represents the temporal functional area loss due to wetland mitigation.

Wetland Valuation

Since we are thinking about temporary losses, the colored areas under the curves in Figure 4 can be thought of in the abstract terms of 'acre-years', which represent a proxy for the total function lost over a given amount of time. As a result, *A*+*B* in Figure 4 represents the temporary functional area loss due to wetland mitigation. This is similar to the manner in which Tong et al. (2007) use value (price) per acre, per year to calculate ecosystem service value of urban Sanyong wetlands in Wenzhou, China. Understanding the number of years that functions are depleted in the landscape is necessary for understanding the total cost of temporary losses. However, calculating cost precisely is very difficult.

Commonly, wetland valuation studies have used conjoint choice, contingent valuation, or other methods for determining the monetary values of wetland resources (Doss and Taff 1996; Mahan et al. 2000; Champ 2003). However, Woodward and Wui (2001), Mahan et al. (2000), and Boyer and Polasky (2004) have shown that these values tend to vary widely and are often quite inaccurate. In their study of Chinese wetlands, Tong et al. (2007) find a potential value of \$7,158 (2007 US\$ or 55332 Yuan) and an actual value of \$751 (5807 Yuan; 10.5% of the full value due to restoration requirements) per acre per year at the Sanyong urban wetland complex. Here, they base their assumptions on the widely cited study on world ecosystem service valuation by Costanza et al. (1997). This ten-fold difference demonstrates the sensitivity to wetland quality that can present in wetland value calculations. For this study, I will calculate low aggregate values from wetland loss using a unit value of \$500 (1993 US\$[3]) per acre per year, and high aggregate values using a unit value of \$10,000 per acre per year.

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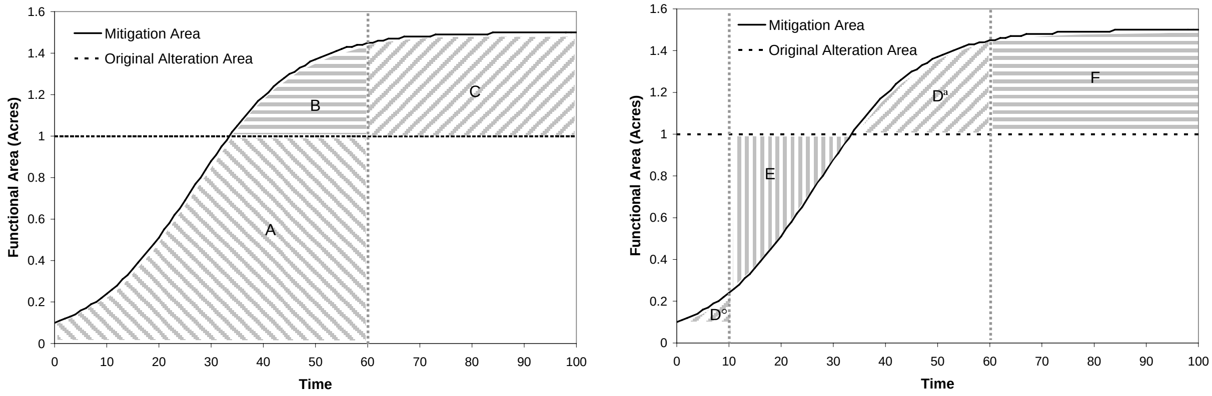
[3] I use 1993 as the normalization time since this is the beginning of the study and simulation period. This figure could be easily adjusted using a Consumer Price Index multiplier.

Method Specific Impact and Mitigation Models

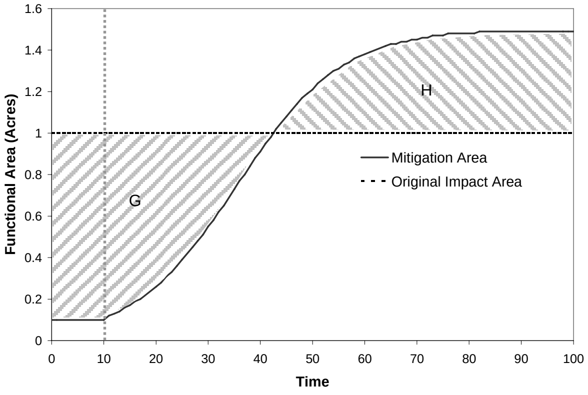
Given these assumptions for the mitigation process, it is possible to analyze several situations that simulate the lag effects associated with ILF and bank mitigation. We can begin this by thinking about a 'theoretical bank' (using the language originally defining banks) in which all mitigation has been performed prior to bank establishment (Figure 5a). Here, bank construction begins at $t = 0$ and a wetland is altered at $t = 60$. During the intervening 60 time periods, the bank establishes a high enough level of functional area that the wetland impact is already offset by the bank before it occurs. As a result, there is no temporal net loss of wetlands, and we actual observe a net temporal gain of wetland functional area since the bank provided functional area alongside the original wetland. This temporal net gain is denoted by the entire shaded area ($A + B + C$) in Figure 5a, where area A denotes the net gain from the growth of the mitigation area to the same level of the original wetland (preceding impact). Area B denotes the gain in wetland function that occurs when the mitigation wetland exceeds the function of the wetland that it is meant to offset (due to 1.5:1 ratio). Finally, area C is the net gain that is carried into the future after the wetland impact at $t = 60$.

Figure 5: Restoration Dynamics for Wetland Mitigation Banks and ILF Sites



A. Theoretical Bank

B. Realistic Bank



C. In-lieu Fee Site

In order to model the more realistic behavior of banks, we can adjust the time at which impacts occur relative to bank establishment.

Figure 5b shows a bank, established at $t = 0$, which begins acting as mitigation for impacts at $t = 10$. As a result, the initial net gain in wetland function (D^o; $0 \leq t \leq 10$) is quickly offset by the net loss caused by a wetland impact (or set of impacts E; $10 \leq t \leq 30$). Although this net loss is eventually offset again by the larger size of the bank relative to impacts (D^a), this does not occur until $t = 60$. Here, we witness a temporary net loss of wetlands in the landscape between $t = 10$ and $t = 60$, after which a net gain of wetlands continues into the future (area F). Although the bank still provides some mitigation prior to impacts, it does not occur early enough to completely overcome the lag effects of wetland impacts.

Finally, we can analyze the case of ILF mitigation, in which fund pooling behavior delays the initiation of restoration activities. In certain areas, this added lag can be quite significant. While ELI (2002) documents the lags associated with ILF programs across the U.S., it draws particular attention to the DuPage County, IL program (see Data Section), which allows the collection of ILF funds for up to ten years prior to beginning restoration work.

We can simulate this behavior by introducing a discrete delay into the initialization of restoration projects.

Figure 5c demonstrates the pattern of temporary wetland loss associated with ILF sites. Here, the initial loss is exacerbated since there is a ten year delay ($0 \leq t \leq 10$) after the impact ($t = 0$) before wetland restoration begins. As a result, area G denoting wetland functional area loss is not offset by gradual functional gain of the ILF site (area H) until well after $t = 100$. This behavior implies that a small lag in starting mitigation can produce a significant extension in temporary net losses.

Although these examples highlight the reference behavior associated with wetland impacts and different mitigation methods, we have only applied this thinking to the case of one wetland impact being mitigated in a bank. In reality, many wetland impacts occur every year and each applies one of the three available mitigation methods. In order to better understand the impacts of wetland mitigation lags, it is important to simulate the actual stream of wetland alterations permitted throughout the landscape and the subsequent mitigation efforts undertaken to offset wetland losses.

Modeling a Series of Wetland Impacts

I begin looking at an extension of the model shown in Figure 2 that simulates a continuous string of impacts occurring within the landscape. It is first important to maintain the structure and behavior associated with wetland impacts (draining net wetland function in the region) and mitigation efforts (gradually increasing net wetland function). In order to do this, I give each year of the simulation an index, whereby impacts and mitigation efforts during each year are represented as individual, but identical model structures[4]. Since mitigation is initiated for new impacts on an annual basis, this array structure now represents a set of logistic growth functions, all beginning during different years. As a result, each year's progress increases a respective array of *mitigation area* stocks, which are summed to represent the functional area gain offsetting impact losses. For each set of impacts, growth rates can be altered based on the relative effort applied to restoration activities. This yields the new model shown in

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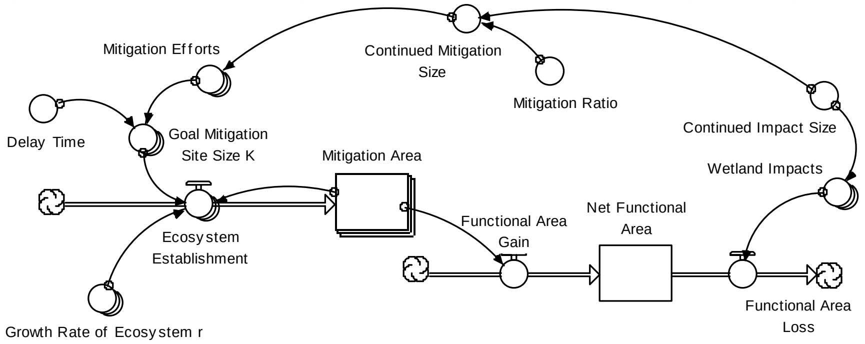
[4] This is performed using the built-in array functionality of the STELLA software. For more information on STELLA, see http://www.iseesystems.com.

Figure 6.

Figure 6: Wetland Mitigation Dynamics Model using Chicago Data



Although impacts occur continuously throughout the year, data on the exact date of impacts are commonly not available (BenDor et al 2007). As a result, impacts are grouped together at the beginning of each of their respective years and simulated as a discrete set. This is represented by a step function that essentially “turns on” a loss of functional area. This function simulates the increase in the functional area loss rate caused by each yearly set of impacts. Here, the behavior is similar to the single loss of functional area shown in

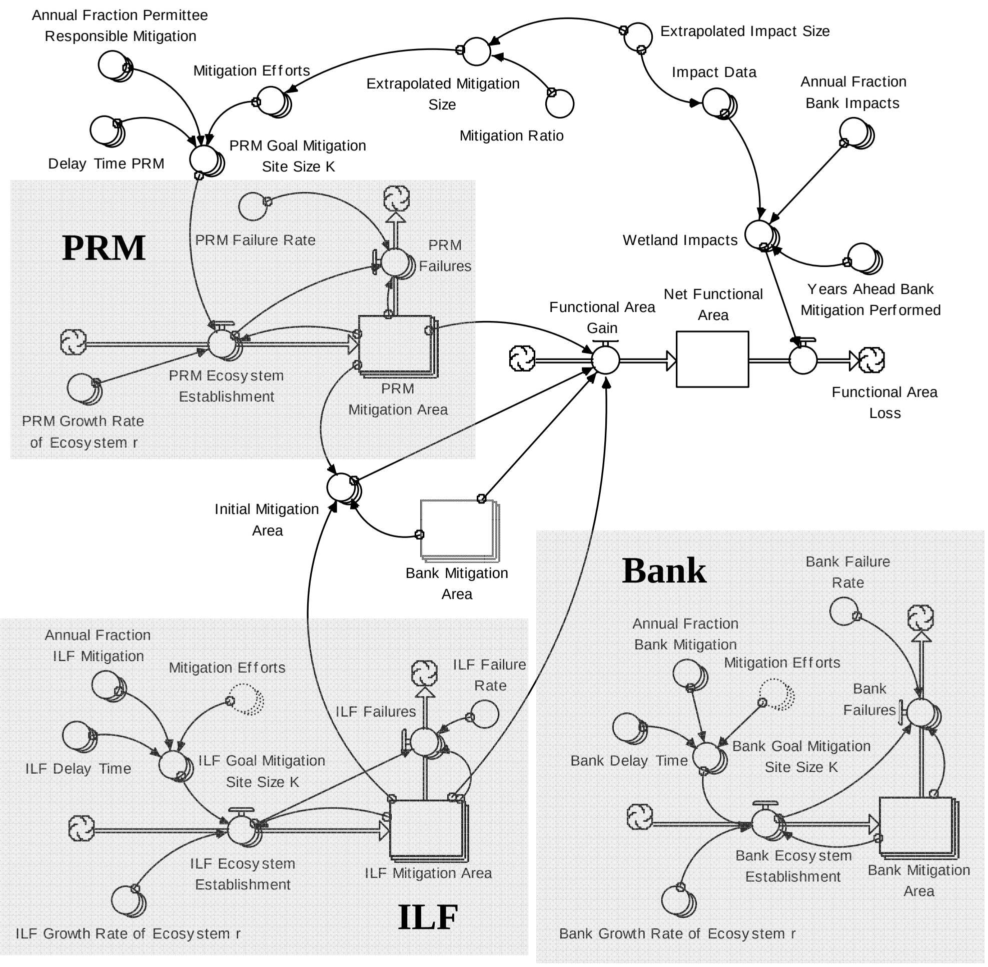
Figure 5, although now, losses compound with new impacts and gains compound with new mitigation efforts. In order to simulate the period after 2004, I estimate an impact size and mitigation ratio that generates new impacts for the remaining time horizon of the simulation. Mitigation data is fed into the model in a similar manner as impacts. After the 1993 – 2004 period of data availability, goal mitigation acreage is determined as the product of *continued impact size* and the *mitigation ratio*.

Disaggregated Mitigation Methods

Although this model effectively simulates a stream of wetland impacts, it does not account for the manner in which those impacts are mitigated, as well as the specific delays that occur with each mitigation method. We can expand this model to incorporate each mitigation method by separating mitigation acreage into individual method-based structures. The model shown in

Figure 7 takes into account the delays associated with starting ILF restoration projects, as well as the pre-impact nature of wetland bank mitigation.

Figure 7: Disaggregating Mitigation Methods



In order to do this, I use the acreage-weighted fractions associated with mitigation projects starting each year to delineate mitigation that uses each method. Here, ILF mitigation contains a discrete delay (*ILF Delay Time*) associated with the start of restoration activities. Likewise, mitigation banking incorporates a delay, although it is applied such that mitigation is actually initiated before impacts. Here, I apply a delay to the occurrence of impacts that use mitigation banks. Although this structure does not accurately portray impact timing, the delay allows for the correct calculation of the relationship between impacts and bank mitigation.

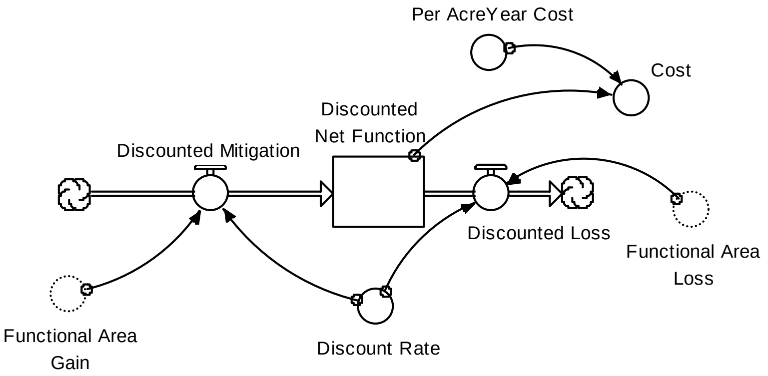
In their discussion of the use of discounting in creating more legally and ecologically defensible mitigation ratios, King and Price (2004) and Gutrich and Hitzhusen (2004) discuss the integration of human time preferences for wetland services and values into a regulatory program. This type of functional time preference can be integrated into the wetland mitigation model as shown by the *discount rate* variable in

Figure 7. In observing the final states in

Figure 5, we see that wetland gains through mitigation may be able to offset wetland impacts and eventually lead to a net gain in functional area. However, the temporary net loss experienced during this process may still be quite extended. As a result, a discounting function will value early wetland gains or losses more heavily than gains or losses experienced far into the future. Here, I apply an exponential discounting function to the functional area gain and loss values (

Figure 7) while calculating the total cost of this temporary loss using an estimated parameter for the per acre, per year value of wetland functions given in Tong et al. (2007). Here, the stock is observed at a distant equilibrium point in the simulated future as discounting term's convergence on zero overwhelms the growth or decline of net function.

Figure 8: Temporary Net Loss Calculation Using a Temporal Discounting Term



This model substructure is identical to the cost calculation given by:

$$\text{Cost} = c\int_{t_0}^{\infty}\left[\left(f_g - f_l\right)e^{-r(t-t_0)}\right]dt \qquad \textbf{(2)}$$

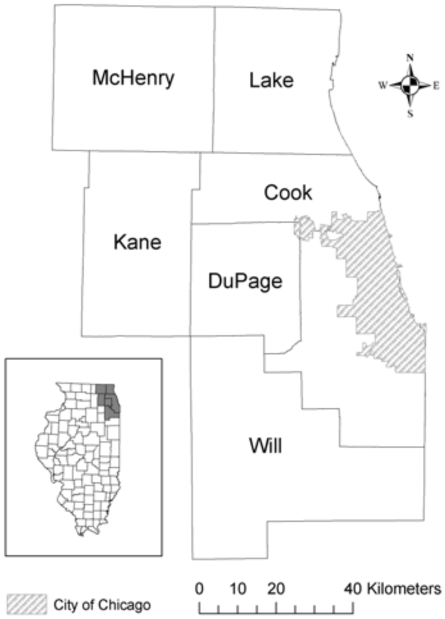
where c is the per acre-year wetland value that is forgone, f_g and f_l are the function gained and lost, respectively, and r is the discount rate.

I will apply these ideas to empirically gathered data for the Chicago region in order to understand the application of regulations and available mitigation methods.

Data

The six-county Chicago region encompasses the Chicago District of the U.S. Army Corps of Engineers, and is home to a complex web of mitigation regulations (Figure 9). This is due in part to Chicago's abundance of wetlands, as well as the region's long history of wetland conversions (Cronon 1991; Robertson 2004). In 1986 and 1987, major flooding in the region convinced the Illinois State Legislature to enact legislation authorizing DuPage, Kane, Lake, McHenry, Will, and Cook Counties to prepare and fund storm water management plans, programs, and projects (Metropolitan Water Reclamation District of Greater Chicago; MWRDGC 2005). Under these storm water management programs, ordinances requiring permits for wetland alterations were established in DuPage (in 1994), Lake (in 2001), and Kane (in 2002) counties. Although the Kane and Lake County ordinances cover only wetlands not under the Corps' jurisdiction (Freeman and Rasband 2002), the Corps has granted DuPage County regulatory authority over all wetlands within County boundaries.

Figure 9: Chicago Region Study Area



The region has attracted more than two dozen active, sold-out, or proposed mitigation banks. Between 1999 and 2001, the Corps also operated an ILF program with a nonprofit restoration organization. Permits were included in this dataset if they were granted between 1993 and 2004, required compensatory mitigation, and included specific data on location and acreage for both alteration and mitigation sites. As a result, this dataset comprises a nearly a census of all completed mitigation transactions in the region between 1993 and 2004.[5]

This dataset is used to gather the acreage of wetland impacts and mitigation attempts for each year between 1993 and 2004. Using information on mitigation method and acreage required for each impact, I am able to find the fraction of mitigation that uses each of the three major mitigation methods, including PRM, ILF mitigation, and wetland mitigation banking[6].

Figure 10: Chicago Impact and Mitigation Acreage by Mitigation Method (1993-2004)

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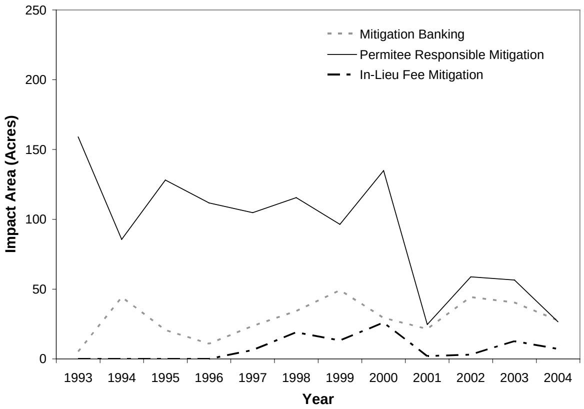
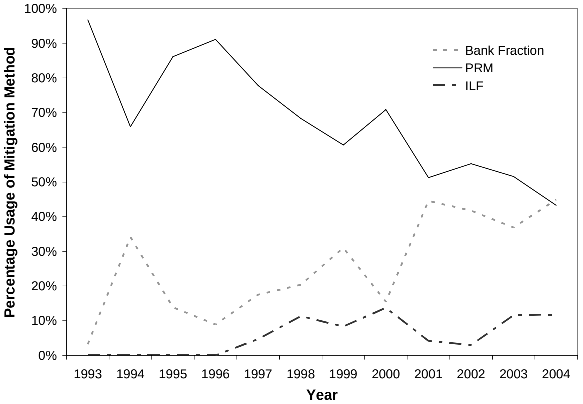
[5] For more information on data sources, collection techniques, and the specific structure and composition of this dataset, see BenDor and Brozovic (2007) and BenDor et al. (2007a).

[6] Each of these fractions is calculated by using the acreage employed by each mitigation technique during a given year.



A. Impact Acreage by Mitigation Method

B. Mitigation Acreage by Mitigation Method



C. Percentage Use of mitigation Method by Acreage

Figure 10a and b shows trends in wetland impacts and subsequent mitigation projects throughout the study period. Although a thorough discussion of the driving forces governing these trends is given in BenDor and Brozovic (2007)[7], it is important to note that there is a strong increasing trend associated with mitigation bank usage that is coupled with the decreasing usage of PRM **(**

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[7] Since these are acreage-weighted trends, the actual usage statistics are partially masked

Figure 10c). Given the potential of mitigation banking to reduce temporary net wetland loss, this trend may have profound implications for wetland development and mitigation over the next several decades.

With these data available, it is now possible to begin thinking about mitigation as a set of responses to a relatively continuous string of impacts and begin modeling restoration behavior accordingly.

Scenario Testing and Results

This section illustrates the testing and simulation of different impact and mitigation scenarios using the model shown in

Figure 7. The first is the 'base case' scenario, where wetland impacts and mitigation projects follow the 1993-2004 dataset (shown in

Figure 10) and are extrapolated at 150 acres of impacts per year, which are mitigated at a 1.5:1 ratio[8]. Here, I make several simplifying assumptions including fairly rapid restoration under all mitigation methods (logistic growth rate of 30%), as well as no ILF delays or bank lead time. I also assume a discount rate is 5% and a per acre-year cost of \$500 to \$10,000 (1993 US\$) when calculating the cost of temporary net losses (per the range given in Tong et al. 2007).

Figure 11 shows the resulting dynamics associated with wetland alterations and mitigation, as well as their impact on net wetland functional area. Impacts decrease functional area for the duration of the simulation, thereby causing a strong initial decrease in net functional area. As mitigation areas establish themselves and grow, total functional gain (the sum of all mitigation for all impacts) exceeds functional loss from impacts, and net functional area begins to grow. Here, the minimum value in the net functional area curve represents to the maximum sum of the differences between functional loss and functional gain. This point is reached at *t = 2037* (line *A*).

Figure 11: Base Case Scenario

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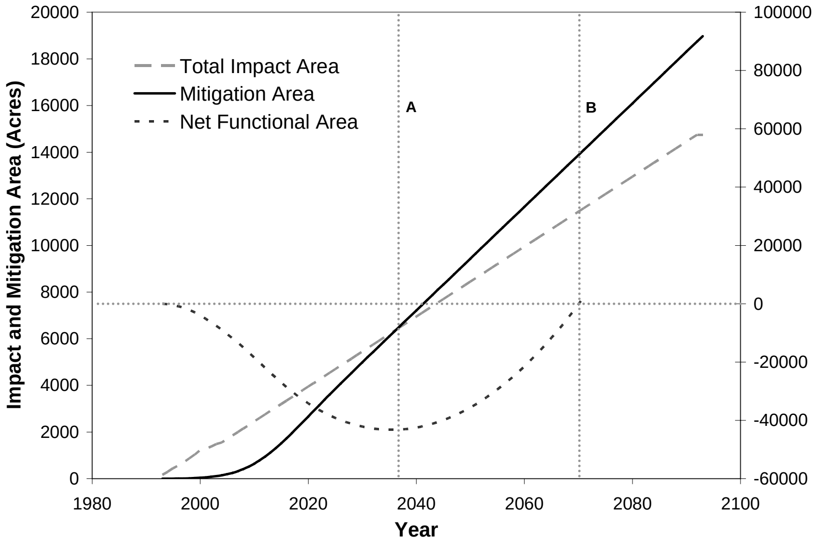
[8] Although it is unreflective of the trends observed in

Figure 10, impacts and mitigation acreage are assumed to be distributed evenly between the three mitigation methods. 150 acres is assumed to be a reasonable annual impact area for the Chicago region given past values (

Figure 10).



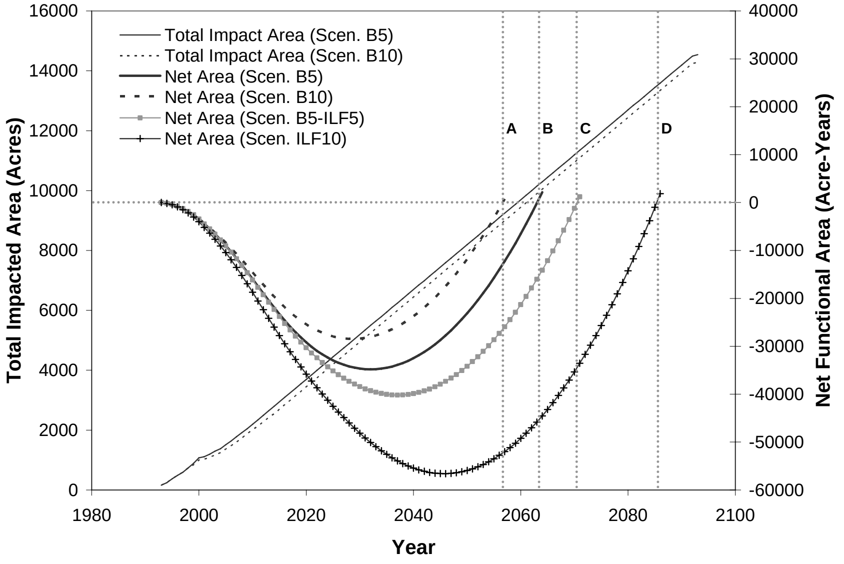
Note: Net Functional Area is given on the right y-axis, while impact and mitigation areas are given by the left.

Net functional area continues to rise as gains from mitigation efforts exceed losses from impacts (line *B*; given on the left-hand side y-axis of Figure 11). Under this scenario, this means that a continuous stream of wetland impacts yield a net functional area loss that is not offset my mitigation efforts until 2071, a time lag of 79 years since the first measured impacts. Given my prior assumptions, I calculate the cost of this prolonged 'temporary' net loss to be between $7.4 million and $74.0 million.

Method-Specific Delays

Next, it is important to highlight the impact of before-the-fact bank mitigation and after-the-fact ILF mitigation. Here, I compare three simulations involving the same impact extrapolation (1993-2023) that have different mitigation 'head starts' for banking and delays for ILF mitigation.

Figure 12: Mitigation Banking and ILF Scenarios



Scenario 'B5' alters the previous assumptions by assuming a bank lead time of 5 years, while Scenario 'B10' assumes a lead time of 10 years. Both of these scenarios assume no delay on ILF mitigation. Here, mitigation remains unchanged in both scenarios ('Functional Gain' curve), and is a result of the assumption that lead time actually delays impacts relative to mitigation. However, the net functional area increases slightly faster in Scenario B10, with mitigation canceling out loss from impacts at time $t = 2057$ (line A) rather than $t = 2064$ (line B) in Scenario B5. However, in both scenarios, early functional losses still cause significant temporal delays, exacting a cost of $5.59 – $55.86 million in Scenario B5 and $4.17 – $41.73 million in Scenario B10.

Given this behavior, we can turn our attention to Scenario 'B5-ILF5', where bank lead time again is set at 5 years, but now ILF delay is also set at 5 years. In this case, functional gain is slowed by the ILF delays, thereby exacerbating the delays seen in Scenario B5 over seven years (Figure 12). Here, mitigation finally offsets impacts during $t = 2071$ (line C) with temporary losses valued at $6.61 – $66.12 million.

A more extended delay time may not be unrealistic; a ten year delay time is currently allowed by the DuPage County, IL storm water management ordinance (ELI 2002; DuPage County 2006) and may already have been exceeded in certain instances (BenDor and Brozovic 2007; BenDor et al. 2007). In Scenario 'ILF10', when ILF delay time are increased to ten years while bank lead times are set at zero, we see a huge increase in temporary losses, with impacts finally offset at $t = 2086$ (line D), yielding a temporary net loss of $9.23 – $92.23 million.

These runs demonstrate the major impact that relatively small shifts in the relationship between impacts and mitigation can have on the aggregate behavior of net functional wetland area. We can now shift our attention to other key factors, including the wetland functional growth rate.

Sensitivity to Wetland Growth Rate

Wetland growth rates can be thought of the speed at which restoration activities are accomplished. Although much more sophisticated methods of assessing functional growth have been established over the years (Brinson 1996), for simplicity I characterize this rate as a single number that generates a single functional growth curve. Unfortunately, this simplification sacrifices the numerous functions whose establishment curves may have much more interesting and nuanced behavior (non-logistic or even non-continuous).

Here, I simulate three scenarios where wetland growth rates are set at 30%, 40%, and 100% per year, respectively.[9]

Figure 13 shows the dramatic changes that wetland growth characteristics can have on net functional area growth.

Figure 13: Wetland Growth Rates during Restoration (30-100%)

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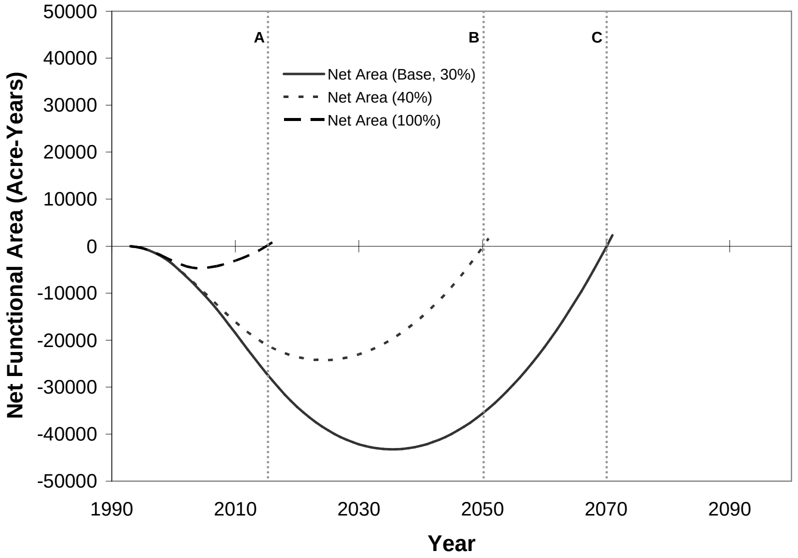
[9] Again, given the logistic nature of this growth, these rates are fractions of the wetland area that have already been established. This leads to exponential growth followed by goal-seeking growth.



Major discrepancies now appear between the times at which net functional area returns to zero. Given a wetland growth rate of 30% in the base case scenario, wetland gains offset losses at $t = 2071$). With higher restoration rates of 40% and 100%, net function is restored at $t= 2051$ and *2016* with costs ranging from \$3.31 – \$33.12 million and -\$6.56 to -\$65.77 million, respectively. Negative costs experienced under a 100% restoration rate actually signal a gain an immediate net benefit from rapid restoration adding to the wetland resource base. Although these increases in restoration rates have major ramifications for temporal net loss, they represent extensive improvements in restoration activities and technology. The calibration of these rates is highly dependent on the region that we are modeling, the functions that we are focused on restoring, and the types of wetlands altered during development. According to Kusler (1990), the restoration of marsh vegetation may take only a few years, while creating wooded swamp land may take decades. Likewise, the buildup of peat and other types of heavily organic soil may take thousands of years. Further studies by Zedler and Callaway (1999), Mitsch and Wilson (1996), and Gutrich and Hitzhusen (2004) have shown that restoration success likely takes much longer than the five year time allotted by regulators (Corps and EPA 1990; Corps 2004.

Given that reasonable restoration times for emergent marsh areas in Northeastern Illinois may range from 5-20 years, an $r=100\%$ may not be unreasonable. Here, full wetland restoration occurs within 15 years of the impact triggering mitigation (Figure 3).

Effect of Increases in Mitigation Ratio

We can now focus on the impact of mitigation ratios on temporal net loss dynamics. Since I already have data describing mitigation requirements for wetland impacts between 1993 and 2004, I look at the effects of changing mitigation requirements for the extrapolated impacts between 2005 and 2093. Here, impacts have a mitigation ratio applied to them that describes the amount of mitigation required to satisfy each permit. This forms the new goal (K) for each year's mitigation efforts. I test three different mitigation ratios, including 0.75:1, in which each acre of wetland alteration is met with 0.75 acres of mitigation, 1.5:1 (the base case), and 4:1. High mitigation ratios are usually reserved for wetlands for which restoration is extremely difficult or entails a high rate of failure.

Figure 14: Changes in Mitigation Ratio for Impacts between 2004 and 2023

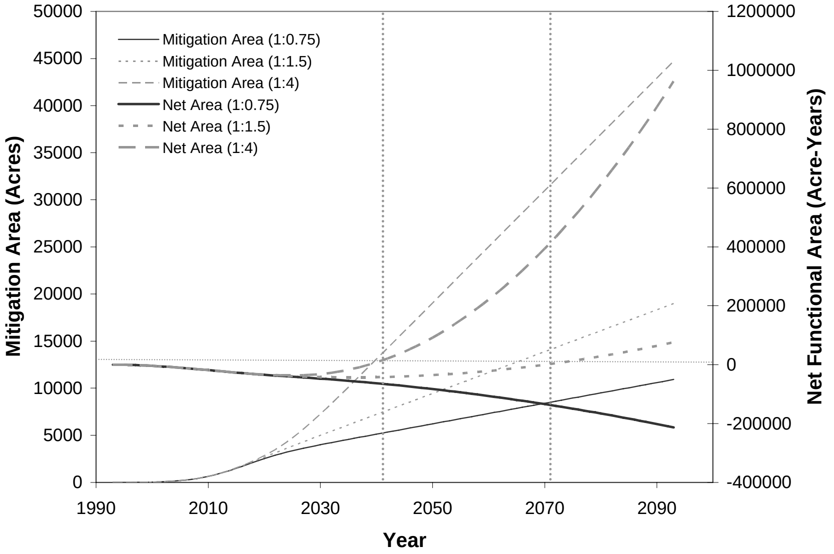


Figure 14 demonstrates that such large changes in mitigation ratios from 1.5:1 to 4 generate extensive impacts on the time at which net functional area offsets losses from previous wetland alteration. Increasing the ratio to 4:1 offsets impacts much more quickly than the base case (*t = 2039*) and decreases costs to between -$8.52 million and -85.22 million. In this case, gains from the high ratio offset temporary losses so quickly that net gains still hold value under the discount function. However, by using a ratio under 1:1, we see that losses are never actually offset by mitigation, leading to a loss of $12.94 to 129.38 million.

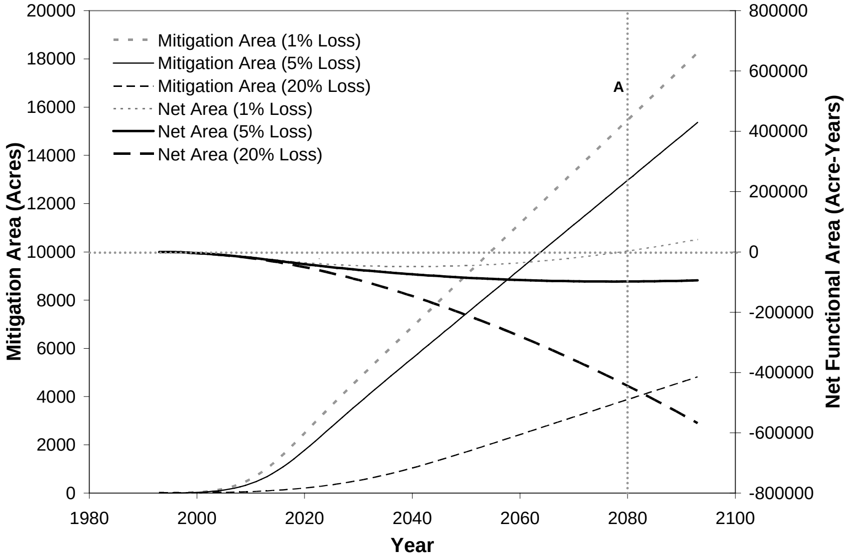
Imperfect Mitigation Performance

A well-studied problem within mitigation is the inability of many mitigation sites to maintain their ecological integrity and yield functioning wetlands (NRC 1992, 2001; see BenDor and Brozovic (2007) and BenDor et al. (2007) for more discussion on ecological criticisms of mitigation). Likewise, wetlands whose functions were restored can eventually succumb to human-induced disturbance since wetland monitoring is often not required for more than five years (ELI 2002). This can result in a decrease in wetland functional area, meaning that net functional area could conceivably dip back below zero, resulting in a second period of wetland losses. As a result, it may be improper to model the mitigation process as a continual accrual of wetland functional area since mitigation often fails.

The model depicted in

Figure 7 has outflows attached to each of the accumulated mitigation area stocks. Each outflow represent a drain on the functional mitigation area attained in every year mitigation is attempted. Here, I simulate aggregate wetland gain given systematically ‘imperfect mitigation’, where imperfect represents a loss of 1%, 5%, and 20% of the accumulated mitigation area during a given year. This behavior simulates a continuing system of poor monitoring and hydrologic, vegetative, and soil substrate establishment.

Figure 15: Imperfect Restoration



As we can see, with only a one percent loss in accumulated mitigation area per year, the time take to offset wetland losses extends eight years (*t = 2079*; line A) beyond that of the base case scenario (Figure 11, *t = 2071*), which assumed perfect restoration. Likewise, simulating five percent and 10 percent restoration losses yield extensions to this offset time of over 30 years, such that wetland losses are not offset within the model’s simulation period.

Discussion

This study has addressed several issues associated with temporal delays in wetland destruction and restoration. First, I have demonstrated that a continuous string of wetland impacts, followed by delays in achieving fully successful compensatory mitigation, can easily lead to an extended period of net loss within the landscape. This occurs even under the assumptions of perfect restoration and mitigation ratios consistently above 1:1. Although significant attention has been paid to accounting systems for assessing wetland losses and gains, dynamic wetland losses have not been addressed consistently by regulators.

This study reveals that poor wetland restoration quality, delays in initiating mitigation, and poor choice of mitigation sites are all capable of creating and exacerbating long-lived restoration delays whose restoration lag costs can easily amount to over $130 million. Moreover, the results show that the rates at which wetland ecosystems are re-established, as well as the relative capability of developers to meet ecological standards, are major determinants of temporary net loss and resulting lag costs.

In any large-scale permitting system, repeated delays in attaining successful mitigation can result in a substantial net loss of wetland function at the landscape level. Given the important ecological and economic roles that wetlands play in urbanizing landscapes, any systematic cause of wetland function loss has significant implications for modern wetland policy.

Conclusions and Policy Implications

Although the 1987 National Wetland Policy Forum made significant contributions to national wetland policy and helped to facilitate a considerable decline in wetland destruction, several recommendations still need to be addressed. While the NWPF may not have intended for new regulations to be responsible for correcting the destruction of previous generations (pgs. 18-19), it is evident that the methods used for accounting for wetland loss and gain in current regulations may not prevent current wetland alterations from burdening future generations.

Acknowledging and protecting against the dynamic, temporary wetland loss has enormous implications for all of the services and values that wetlands provide, including flood prevention, water quality improvement, and wildlife and bird habitat (NRC 2001). As a result, I contend that estimating wetland restoration delays on an aggregate scale is not only possible, it is absolutely necessary for evaluating the extent to which regulatory programs uphold no net loss policy and its intended effects on our Nation's wetland resource base.

Although I have made many simplifying assumptions about impacts and the mitigation process, many of these assumptions mirror those made by regulators in permitting programs at the federal and local levels. By assuming that wetlands can be commodified based on their area alone, regulators may shut out many important wetland quality considerations (Ruhl and Gregg 2001). Although recent studies have made a convincing case for the creation of a more robust currency for wetland trading (Salzman and Ruhl 2000, 2004), the current use of area as a proxy for wetland function can be substantially improved.

While sophisticated systems have been created to evaluate wetland functional equivalence for regulatory purposes (Bedford 1996; Brinson 1996; Lupi et al. 2002), their use directly depends on the time and energy that regulators have to consider, implement, and enforce wetland alteration permits. The prospect of using these techniques for hundreds, if not thousands, of assessments can easily overwhelm regulator resources. As a result, regulators need a system for efficiently and rigorously estimating useful mitigation ratios, promoting the use of rapid mitigation methods, and enforcing stringent siting requirements.

I have shown that restoration lag costs in the Chicago region, an area that has already experienced dramatic wetland loss[10], can be as high as $130 million. This, combined with the high restoration lags costs estimated by Gutrich and Hitzhusen (2004), suggest that eliminating lags should be a paramount priority in new mitigation regulations. Prior studies have recognized

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[10] The Chicago region has lost over 75% of its naturally occurring wetlands due to urban and agricultural development (Robertson 2004).

that wetland restoration is not a static activity, but rather is a dynamic process with a complex set of goals (Hobbs and Harris 2001). Given this, regulators already have many of the tools necessary to lower the type of dynamic, temporary net loss now impacting the landscape. Actions can include lengthening the time for which developers are responsible for monitoring mitigation wetlands and ensuring successful restoration. Additionally, by raising mitigation ratios for development on wetlands with slow re-establishment conditions (such as swamps and other wooded wetlands), as well as tightly enforcing the use of highly viable mitigation areas for all impacts, regulations can significantly raise the rate of functional re-establishment. Using these tools, regulators can ensure that the spirit of the no net loss policy is upheld, where wetland functions are maintained at the landscape level, even if they are not achieved on a permit-by-permit basis.

With the development of more extensive data collection infrastructure on the part of the Corps, it may soon be possible to disaggregate ‘impacts’ into a specific set of wetland functional losses (Olson 2004, 2005). Although the high mitigation ratios that I advocate here may protect the Nation’s wetland base, they also create a greater economic barrier to wetland destruction. This may have the unintended consequence of promoting un-permitted wetland impacts. By collecting better data and creating planning and regulatory support systems (BenDor et al. 2007), authorities may be able to prevent illegal wetland destruction by assisting developers with locating suitable mitigation sites, as well as streamlining the permitting process and reducing costly permitting delays. Likewise, future modeling research on this topic must deliver a more sophisticated representation of wetland functionality. Subsequently, disaggregating ‘mitigation area’ to study the nuanced behavior of processes like floral community succession be incorporated as a powerful tool for more accurately informing the regulatory permitting process.

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