ANALYSIS

Assessing the substitutability of mitigation wetlands for natural sites: estimating restoration lag costs of wetland mitigation

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Abstract

The extent and rate to which mitigation wetlands can replace the functions of natural ones remains uncertain. Further, the economic time lag costs of wetland function restoration and therefore cost-effective and efficient means of wetland mitigation have yet to be adequately addressed. In this study, 16 mitigation wetlands were assessed, comprised of eight low elevation inland freshwater emergent marshes in Ohio and eight high elevation (>2285 m) freshwater emergent marshes in a wetland complex in Colorado, USA. This research identified the ecological substitutability of mitigation inland freshwater marshes for natural ones, estimated economic restoration lag costs to society and addressed least-cost approaches to successful mitigation.

Years required to achieve full functional equivalency for both floristics and soils for the Ohio sites under logarithmic growth ranged from 8 to 50 years with a median of 33 years. Years required to achieve floristic functional equivalency for the Colorado sites ranged from 10 to 16 years with a median of 13 years. Restoration lag costs per acre (0.4 ha) in Ohio ranged from $3460 to $49,811 per acre with an average of $16,640 per acre (2000 US$). Lag costs as a percentage of total restoration costs ranged from 5.6% to 52.8% with an average of 25%. Restoration lag costs per acre to achieve full floristic equivalency in Colorado ranged from $22,368 to $31,511 per acre with an average $27,392 per acre. Time lag costs as a percentage of total restoration costs ranged from 44% to 53% with an average of 49%. Findings of this research suggest that society is currently incurring significant wetland restoration costs due to time lags of mitigation sites. Requiring the posting of an interest accruing performance bond can serve to internalize the time lag costs to the permittee and provide an incentive for more cost-effective wetland restoration efforts.

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Keywords: Valuation; Mitigation wetlands; Restoration lag costs; Environmental policy

1. Introduction

Wetlands are increasingly recognized as valuable natural systems providing useful services to society such as flood abatement, water purification, groundwater recharge, erosion control, and biological diversity (Ewel, 1997). International recognition of the
value of wetlands is apparent through collective action in the Convention on Wetlands of International Importance (Ramsar Convention; Mitsch and Gosselink, 2000). Historical degradation of wetlands in the United States has led to a federal “no net loss policy” for wetlands authorized by the Clean Water Act (Findley and Farber, 1992; National Research Council, 1995). The Clean Water Act mandates the restoration and maintenance of the chemical, physical and biological integrity of the nation’s surface waters including certain wetlands (Fennessy and Roehrs, 1997; Fennessy et al., 1998).

In the state of Ohio, 90% of the original wetlands have been converted over the last two centuries (Dahl, 1990). Under the rules guiding the protection of water quality in Ohio, wetland losses are prohibited without compensatory “mitigation”—restoring or creating a wetland to make up for the one destroyed in the process of development (Fennessy and Roehrs, 1997). The United States Environmental Protection Agency (USEPA) and United States Army Corps of Engineers (USACE) along with state agencies such as the Ohio Environmental Protection Agency (OEPA) regulate and monitor the mitigation process providing specifications for creating a new wetland or restoring a degraded one usually in the same watershed (Marsh et al., 1996; Salvesen, 1993). OEPA studies refer to wetlands created or restored through construction efforts to meet federal no net loss requirements as “mitigation wetlands” (Fennessy and Roehrs, 1997).

Wetland mitigation is viewed as a means to balance the need for economic development with environmental protection (Fennessy and Roehrs, 1997; Shabman et al., 1998). Extensive debate has centered around the (1) ecological criteria of a “successfully” created or restored wetland; (2) the extent to which manmade mitigation wetlands substitute for ecological functions of natural sites and (3) the economic implications of wetland mitigation (Bystrom, 1998; Green and Söderqvist, 1994; Malakoff, 1998; Mitsch et al., 1998; Nadis, 1999; Niswannder and Mitsch, 1995; Shabman et al., 1998; Wilson and Mitsch, 1996; Zedler, 1996a). Wetlands are the only ecosystem to be comprehensively regulated across all public and private lands in the United States (National Research Council, 1995). As a result, wetlands serve as ideal systems for ecological and economic research addressing the ability of society to substitute ecological functions and services of natural ecosystems with manmade replacements.

In an effort to address the need for empirical research on the substitutability of natural capital, this research estimates the ecological substitutability of mitigation wetlands versus natural wetland sites and the effects on economic time lag costs to society (i.e. temporal loss of social wetland benefits). This research consists of two main components: (1) ecological functional assessment of mitigation wetlands versus natural reference sites for eight Ohio inland freshwater mitigation wetlands at low elevation and eight Colorado inland freshwater mitigation wetlands at high elevation and (2) incorporation of functional replacement data into an ecological–economic simulation model that analyzes the ecological functional substitutability of mitigation wetlands for natural ones and the economic restoration lag costs incurred while achieving natural functional equivalency. Implicit in this study is a comparative analysis of historic US classification and assessment schemes with more recent quantitative ecological functional analysis of wetlands.

Ecological economic models that address trade-offs between ecological services and economic costs have been constructed and utilized in socio-economic decision-making, but are rare (Baker et al., 1991; Costanza et al., 1995; Fernandez and Karp, 1998; Grossman, 1994; Odum, 1994). A greater rarity is empirical analysis and modeling of the correlation of economic costs and substitutability of ecological functions (Bystrom, 1998; Costanza et al., 1995; Fernandez and Karp, 1998). In this study, functional and economic data are incorporated into a computer simulation model to assess cost-effective means for restoration through analysis of the total and marginal restoration lag costs of mitigation wetlands. Resource based compensation such as the construction of a wetland to restore lost functions of a natural site inherently contains time lag costs (foregone societal wetland benefits; Mazzotta et al., 1994). Historically, time lag costs have not been considered in cost estimation of wetland restoration (King, 1998; King and Bohlen, 1994). Specifically, our model estimates economic costs correlated to the level to which one can substitute for the natural function of one class of wetlands: inland freshwater marshes. Simulation results provide estimates of (1) years required by...
mitigation wetlands to achieve functional equivalency of natural sites; (2) time lag costs for low and high elevation mitigation sites and (3) percent lag costs of total restoration costs as an indicator of public versus private restoration costs.

2. The economic rationale

As displayed in Fig. 1, the decision process for wetland drainage can be considered as a trade-off at the margin between benefits from development (MB) versus the social costs of natural wetland loss (SMC). Historically, the benefits from draining a wetland and converting it to agricultural land as displayed in the market far exceeded the private costs (PMC) associated with lost natural function of the wetlands. Thus, there existed strong economic incentives for conversion of wetlands and little incentive for preservation of wetlands by the private landowner. In a fully-functioning market with perfect competition and properly estimated benefit and cost curves, one would expect the number of drained wetlands to increase until the point at which marginal benefits derived from converting wetlands were equal to the marginal costs of draining wetlands (see Fig. 1). In the past, the private landowner considered a marginal trade-off between developmental benefits versus private foregone wetland benefits and often chose conversion of the wetlands. Unfortunately, the private landowners often did not consider the full social costs of draining the wetlands which led to an inefficient over-conversion of wetland sites (Dahl, 1990).

Wetlands, like many environmental resources, provide ecological goods and services that are valuable to society but are often not reflected in established markets—termed “non-market” goods and services (Dixon et al., 1994). A private landowner can reap

![Fig. 1. The trade-off between maintenance of wetland function and conversion of wetlands with a no net loss standard (MB, marginal benefits; PMC, private marginal costs; SMC, social marginal costs; P, price; WC, quantity of wetland functions converted). The US “no net loss” policy requires that wetland functions be maintained at current levels either through preservation or construction of wetlands. The no net loss policy serves as a safe minimum standard of wetland protection at current levels ($W_{NNL}$).](image)
benefits from farmland by selling agricultural commodities in a market, but often cannot reap benefits of wetland goods and services that are of high social value for which no market exists. For example, wetlands provide ecological goods and services such as flood protection, water quality improvement, aquifer or groundwater storage and recharge, buffers against erosion, sources of wildlife habitats, landscape amenities, biodiversity, and carbon sequestration (Turner, 1992). Yet most of these goods and services historically have not been considered in the decision-making process. Only loss of commercial applications and products of wetlands such as fish, furs, waterfowl, peat fuel, low-intensity grazing and recreational opportunities have been considered as foregone costs by private landowners (King, 1994; Turner, 1992). Thus, the costs of lost services from natural wetlands have been underestimated and the social costs of wetland conversion may exceed private costs (SMC>PMC; see Fig. 1). If social costs of wetland drainage are higher than private costs, efficient use of wetlands would require less wetland conversion. The extent to which wetland conversion should be curbed is dependent upon the social value of wetland goods and services.

Over the last 30 years, economists have increasingly attempted to estimate the full social value of environmental goods—quantified as an estimate of the willingness-to-pay for both marketed and non-marketed environmental goods and services. Valuation techniques including hedonic pricing, travel cost methodology, contingent valuation methods (CVM) and choice modeling have been applied to assess the non-market value of environmental goods and services (Blamey et al., 1999; Boyle and Bishop, 1987; Brookshire et al., 1983; Randall et al., 1983). Numerous studies have specifically attempted to provide estimates of the value of non-market wetland goods and services (Bateman et al., 1995; Bell, 1989; Farber, 1996; Farber and Costanza, 1987; Folke, 1991; Green and Söderqvist, 1994; Lant and Roberts, 1990; Morrison et al., 1999; Thibodeau and Ostro, 1981).

Growing recognition of social benefits from wetland functions contributed to the US mandate of “no net loss” of wetland function. Implicitly, policymakers have indicated that historical drainage has exceeded an efficient level of wetland conversion and thus have established a minimum standard to maintain social wetland benefits. This US standard can be represented as a set amount of required wetland functions as indicated in Fig. 1. Failure to consider the social benefits of wetland functions (i.e. social costs of destroyed wetland functions) would lead to further degradation of wetlands to an inefficient level of WC1. If social benefits were incorporated in the decision-making process, efficient wetland use would require maintenance of wetland function at a lower level of wetland conversion indicated as WC2. The level to which wetlands should be maintained is contingent upon estimation of the social benefits of wetlands.

Although numerous studies have been completed, most research focuses on a few wetland functions and thus represents low bound estimates of the social value of wetlands (King, 1998). The uncertainty of the social value of wetlands has led to extensive debate as to the efficient level of wetland use (i.e. is the efficient level = WC1, W_NNL, or WC2?). Although establishing a no net loss policy does not guarantee efficient use of wetlands, at minimum, it establishes a standard that ensures maintenance of social benefits of wetlands up to the level of the standard when markets fail to consider these benefits in private decision-making. Determining more cost-effective means to maintain these social wetland benefits captured by the no net loss standard is a focus of this research. Further, low bound estimates of social foregone wetland benefits are utilized to more accurately reflect the time lag costs of wetland mitigation to society and promote more economically efficient use of wetland resources.

In order to meet the no net loss requirement, a social planner (i.e. USACE, USEPA) must address the trade-off between the social costs of constructing mitigation wetlands versus the costs of preserving the natural site (i.e. foregone opportunities of development). Due to the fact that development opportunities are often reflected in markets and the social benefits of wetlands are often not quantified, the arguments for development often favor drainage of the natural site and mitigation proceeds by constructing wetlands. In the mitigation process, it is assumed a priori that mitigation wetlands will serve as “in-kind replacement” and restore all the functions of the natural site. However, there are numerous examples of mitigation projects that have failed to replace functions of the natural site over time (Erwin, 1991; Kentula et al., 1992; Malakoff, 1998; Nadis, 1999;
The extent and the rate at which mitigation wetlands can substitute for the functions of natural sites remains uncertain. From an economic efficiency standpoint, it is unclear whether the \textit{a priori} decision to rely on the mitigation wetland to meet the US requirements incurs less social costs than preserving the natural site. Implicitly, construction costs and time lag costs (i.e. temporal loss of social wetland benefits) rise with increased difficulty of substitution of a wetland function and one would expect wetland preservation in the case where restoration (e.g. substitution) costs exceed foregone opportunity costs of development. Conversely, functions that are readily substituted would incur relatively low construction and time lag costs and may indicate building more cost-effective mitigation wetlands. Current practice is to require mitigation area ratios >1 to allow for possible failures.

\textbf{Fig. 2} displays potential cost curves for natural and constructed sites when a decision has been made to rely on the mitigation wetlands to provide the wetland functions. Under an \textit{a priori} assumption that mitigation wetlands will provide full functional replacement, the choice to preserve the natural site becomes a static decision at time zero and often is made by simply comparing the development benefits to the private cost of mitigation. Only those projects that display high private restoration costs and/or high “quantifiable” social restoration costs at time zero (i.e. exceeding foregone benefits from draining the natural site) will result in a decision for preservation (TSRC\textsubscript{11} in Fig. 2). Unfortunately, full functional replacement is not assured and the time lag for the constructed site to replace the functions of the natural site drives up the social opportunity costs of foregone wetland benefits. Further, if an agency determines that a mitigation effort is failing it may require further mitigation efforts (i.e. more seeding, contouring, etc.) that increase restoration costs. The level and rate at which one can substitute mitigation wetlands for natural sites significantly influences social restoration costs. Thus, an \textit{a priori} decision to utilize mitigation wetlands to meet the no net loss requirement may be economically inefficient if the total restoration costs (i.e. private

\begin{figure}[h]
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\includegraphics[width=\textwidth]{fig2.png}
\caption{Graphic representation of total restoration cost curves when an \textit{a priori} decision has been made to rely on mitigation wetlands to provide the required level of wetland function. Restoration costs (TPRC, total private restoration costs; TRC \textsubscript{w/lags} = private restoration costs + restoration lag costs to society; TSRC, total social restoration costs; TSRC\textsubscript{11}, high total social restoration costs) are correlated to the rate at which mitigation wetlands can obtain natural functional equivalency (no net loss) over time. The \textit{a priori} decision to utilize constructed sites is inefficient when restoration costs > benefits from development before obtaining functional equivalency.}
\end{figure}
mitigation costs + social opportunity costs) exceed the costs of preserving the natural site before reaching the no net loss US standard (see Fig. 2). Utilization of mitigation area ratios >1 attempts to provide additional social benefits from additional wetland acres (discounted) that may offset time lag costs incurred while waiting for full functional restoration. However, more acreage does not necessarily ensure more ecological functions restored.

In the case of more perfect production substitutes, one would expect that mitigation wetlands can restore functions of natural sites more rapidly within a time frame of natural establishment of communities, supporting the argument for mitigation efforts (see Fig. 2). In the case of perfect production complements, SRC would increase rapidly as mitigation wetlands fail to produce functions equivalent to natural sites (represented by the Y axis) making a case for preservation (see Fig. 2). Only empirical analysis will reveal the rates at which mitigation wetlands can substitute for these natural functions and provide insight as to the optimal strategy for achieving no net loss of wetland functions in an economic least-cost and more efficient manner.

3. The ecological–economic model

In this study, ecological and economic data are integrated in an ecological–economic model that estimates the total restoration costs including time lags of achieving the required objective of no net loss of wetland function. Restoration lag costs are a function of ecological rates of wetland substitution and decrease with the increased ability of a mitigation wetland to restore all the functions of a natural site quickly. Once wetland mitigation has been chosen over preservation, the objective is to minimize total restoration costs with time lags while achieving natural functional equivalency. This can be represented as:

\[
\text{Min } \text{TRC}_{w/lags} = \text{FRC} + \int_0^t \text{VRC}(t)e^{-rt}dt + \int_0^t \text{RC}_{lag}(N(t), \text{FWS}(t_0)) - \text{FWS}(t)e^{-rt}dt
\]

where \(N=f(s, t)\); subject to \(N(t) \geq N^*\); \(\text{TRC}_{w/lags}\), total restoration costs with time lags; \(\text{RC}_{lag}\), lag costs of...

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Fig. 3. Conceptual schematic of the ecological economic model indicating key variables, functions, and functional indicators of the mitigation wetland (OM, organic matter; symbols after Odum, 1994).
foregone wetland goods and services; FRC, fixed construction costs of a mitigation site; VRC, variable costs (monitoring, additional seeding, etc.); FWS, foregone wetland goods and services; \( N \), level of functional indicator of the constructed site; \( s \), rate of functional restoration; \( r \), discount rate; \( t \), time; \( N^* \), average level of natural reference functional indicator (i.e. functional equivalency).

Fig. 3 displays a conceptual schematic of a wetland ecosystem indicating the link between key wetland processes and chosen functional indicators as success criteria of the mitigation wetland. Model simulations were run for the functional indicators of (1) plant species richness, (2) percent native plant species, (3) floristic quality (as indicated by the Floristic Quality Assessment Index, FQAI), (4) percentage hydrophytes and (5) percent soil samples with colors indicative of hydric soils. Simulations were run over a 50-year time period with the Runge-Kutta-4 integration method (DT = 1). Natural equivalency was achieved once the functional indicator of the constructed site was greater than or equal to the average level of the functional indicator for the natural reference wetlands (see Table 1).

Private restoration costs were estimated at $57,885 per acre (0.4 ha; 2000 US$) based on findings of King and Bohlen (1994). King and Bohlen estimated wetland restoration cost per acre for freshwater emergent marshes at $48,700 (1993 US$) from a primary data set of 28 wetland mitigation projects that displayed attention to pre-construction research and post-construction maintenance ensuring high likelihoods of meeting restoration targets. Thus, this cost estimate reflects average expenditure per acre (excluding land costs) for the “best efforts” for inland freshwater marsh restoration towards achieving restoration goals. King (1998) has made the argument that such cost estimates may serve as a “revealed willingness-to-pay by policymakers” for wetland restoration through in-kind replacement. In this analysis, the amount of $57,885 per acre is utilized to estimate time lag costs to society as a low-bound estimate for foregone wetland goods and services.

Private restoration cost estimates were obtained for JMB ($14,310 per acre) and Gavin ($34,959 per acre) from consulting reports and public releases (Dempsey, 1996; Law Environmental, 1992). For the high elevation sites, costs for the 8- and 14-year sites were

<table>
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<th>Table 1</th>
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<td>Functional indicators, restoration over time and success criteria of the mitigation wetlands</td>
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<tr>
<td>Functional indicator</td>
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<tr>
<td><strong>Low elevation sites (Ohio)</strong></td>
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<tr>
<td>(1) Plant species richness</td>
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<tr>
<td>(2) Percent native plant species</td>
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<tr>
<td>(3) Floristic quality (FQAI)</td>
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<tr>
<td>(4) Percent hydrophytes</td>
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<td>(5) Percent soil samples w/ hydric soil colors</td>
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<tr>
<td><strong>High elevation sites (Colorado)</strong></td>
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<tr>
<td>(1) Plant species richness</td>
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<tr>
<td>(2) Percent hydrophytes</td>
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<tr>
<td>(3) Percent soil samples w/ hydric colors</td>
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The FQAI is a ranking index of floristic quality and is calculated as follows: 

\[
I = \frac{1}{N} \sum_i R_i 
\]

\( I \), the FQAI score; \( R \), sum of coefficient of conservatism (C of C) and \( N \), total number of native species. Coefficients of conservatism for each plant species are estimated on a ranking scale of 1-10 with values assigned under the following criteria (Fennessy and Roehrs, 1997): values of 1–3: taxa that are widespread and not indicative of a particular plant community; values of 4–6: species typical of a successional phase of a native community; values of 7–8: taxa typical of stable or “near climax” conditions; values of 9–10: taxa exhibiting high degrees of fidelity to a narrow set of ecological parameters.

\( a \) Mean values ± standard error.

\( b \) Functional equivalency based on the average value observed at the natural reference sites at both low and high elevations.

\( c \) As presented in Andreas and Lichvar (1995); Fennessy and Roehrs (1997).
estimated from consulting reports for a newly constructed 1999 wetland site. The same consultant that was involved with the construction of the 8-year site reported wetland restoration costs at $28,565 per acre (2000 US$; Bio-Environs Consulting, 1999). For the rest of the low elevation sites, private restoration costs were estimated as the cost required to achieve natural equivalency with a high likelihood at $57,885 per acre. Land costs were excluded from private “restoration” costs in this analysis because often land value is more a function of developmental uses (commercial, agricultural, housing, etc.) captured in private markets than a function of the land attributes that will enhance wetland restoration efforts of non-market ecosystem goods and services (social benefits).

Restoration lag costs were calculated as the difference between the value of the wetland goods and services at the time of draining (i.e. time zero) and the discounted value of wetland goods and services in 1-year intervals of incremental replacement until full functional replacement was achieved. Discounting occurred over 50 years at a rate of 8% for each mitigation wetland. Restoration lag costs were estimated considering the incremental paths of functional replacement under the assumption that a mitigation wetland returning 50% of the wetland goods and services is more valuable than if it replaced 25% at the same point in time even though both scenarios have yet to achieve full functional replacement.

4. Field data collection

In this study, 16 constructed wetlands were assessed, comprised of eight low elevation inland freshwater emergent marshes in Ohio and eight high elevation emergent marshes in a wetland complex in Colorado. For the Ohio sites, collection of field data occurred in the Fall of 1999 representing the second intensive set of ecological functional data collected for each constructed wetland. In 1995, the OEPA conducted functional assessments of these Ohio sites to estimate rates of functional development and preliminary success of wetland mitigation projects (Fennessy and Roehrs, 1997). This research expands upon the 1995 OEPA findings by extending the time series to determine if these eight constructed inland marshes are increasing in their functional capacity towards levels of natural sites over a greater time horizon when analyzing functions from 1–4 years (1995) to 5–8 years (1999). Similarly, eight constructed inland freshwater marsh basins in a high elevation wetland complex in Colorado were assessed from June to August in 1998 and 1999. The same field protocols were utilized to compare ecological functional trajectory after 8 and 14 years of development as well as economic analysis across regions and elevations. Field data were utilized to calibrate the ecological-economic model (in STELLA™) to simulate paths of functional restoration over time and estimate incremental economic time lag costs for each site.

5. Key findings and results

Field results of this study indicate that functional replacement by low elevation mitigation marshes in Ohio has leveled off after rapid replacement in early years (1–4 years after construction) and still remains below equivalency for certain functional indicators after a minimum of 5 years development (see Table 1). Regression analyses suggest logarithmic functional restoration \((y = B\ln(x) + C)\) for the low elevation sites in Ohio due to the rapid replacement displayed in early years with lower rates of functional replacement in latter years (see Fig. 4). Regression results for non-linear logarithmic replacement of floristic functional indicators display a closer fit to the data (species richness: \(R^2 = 0.52\); percent natives: \(R^2 = 0.75\); floristic quality: \(R^2 = 0.61\); percent hydrophytes: \(R^2 = 0.72\)) than linear approximations \((y = mx + b)\); species richness: \(R^2 = 0.20\); percent native species: \(R^2 = 0.19\); floristic quality: \(R^2 = 0.24\); percent hydrophytes: \(R^2 = 0.19\)).

After 1–4 years of restoration, constructed sites displayed significantly lower percent native plant species and percent hydrophytes than natural reference sites. Given another 4 years for functional restoration, the mitigation wetlands were still on average significantly less than natural reference sites for percent native plant species and percent hydrophytes (see Table 1). No significant difference was observed for any floristic or soil indicator when comparing the constructed sites over time in 1995 and 1999. Constructed sites displayed lower average
Fig. 4. (a) Regression analysis of functional replacement (% native species) over time for the Ohio sites, (b) % native plant species restoration over time for each Ohio site and (c) calibration of field data at the Aurora site to the best fit of logarithmic restoration of % native plant species (logarithmic: $y = B\ln(x) + C$; logistic: $y = y_0 e^{rt}/(1 + y_0 e^{rt}/C_0)$; sigmoidal: $y = e^{C - Bt}$).
values, but no statistical significant difference between percent soil samples with hydric colors when comparing 1995 and 1999 results to natural reference sites.

High elevation sites in Colorado indicate that functional replacement is increasing over time for certain indicators with 14-year wetlands displaying significantly greater number of plant species than 8-year wetlands. The 14-year sites displayed significantly lower percent hydrophytes than the 8-year sites, but in aggregate terms still contained more hydrophytes. On average, the 8- and 14-year wetlands have not reached equivalency for number of plant species as displayed in the high elevation natural reference sites. On average, the 8- and 14-year wetlands displayed no significant difference from natural sites for the hydric soil indicator.

Field data for the high elevation sites in Colorado suggested logistic growth \((y = y_0 * e^{rt}/(1 + y_0 * (e^{rt} - 1)/K))\) of species richness \((R^2 = 0.92)\) and logarithmic replacement for percent hydrophytes \((\log \text{arithmic: } R^2 = 0.89)\). In consideration of the field data and regression results, years required to attain natural equivalency and time lag costs for the Colorado sites were estimated by calibrating the model for each site with logistic replacement of plant species and logarithmic restoration of percent hydrophytes. For the Ohio mitigation sites, logarithmic replacement was modeled for all functional indicators: plant species richness, percent native plants, floristic quality \((\text{FQAI})\), percent hydrophytes, and percent soil samples with hydric colors.

Years required to achieve functional equivalency of all floristic indicators in Ohio range from 7 to 44 years with a median of 14 years (see Table 2). Restoration lag costs to society to wait for the mitigation wetlands to achieve full floristic equivalency range from $5190 at the Aurora site to $309,108 at the JMB site with a median of $52,794. Restoration lag costs range from $2939 per acre (0.4 ha) to $11,179 per acre with an

<table>
<thead>
<tr>
<th>Rank</th>
<th>Size (acres = 0.4 ha)</th>
<th>Time lag costsa (2000 US$)</th>
<th>Rank</th>
<th>Time lag costs (per acre)</th>
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<th>Private restoration costs ($)</th>
<th>Total restoration costs w/lags</th>
<th>% Lag costs of total costs</th>
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<tr>
<td>(1)</td>
<td>Aurora 1.5</td>
<td>5190(^{(9)})</td>
<td>(1)</td>
<td>Aurora 2939</td>
<td>(1)</td>
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<td>Rittman 1.4</td>
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<td>Aurora 3460</td>
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</table>

**Total Res. Costs w/lags = Total restoration costs with lag costs = private restoration costs + lag costs.**

\(^a\) (x), years required to attain natural functional equivalency with logarithmic growth; all dollars adjusted for inflation according to the Consumer Price Index, US Department of Labor; PV calculated over 50 years with an 8% discount rate; lag costs are based on functional restoration in annual increments.

\(^b\) Total Res. Costs w/lags = Total restoration costs with lag costs = private restoration costs + lag costs.
average of $6136 per acre. Ranking sites according to percent lag costs of total restoration costs offers an indication of the percentage of restoration cost that society incurs in the attempt to mitigate natural wetland losses. These lag cost percentages range from 5.6% to 25.5% with society incurring an average of 12%

Lag costs increase for the low elevation sites on average when requiring each site to achieve both floristic and soil equivalency (percent soil samples with hydric colors). Years required to achieve equivalency increased ranging from 8 to 50 years with a median of 33 years. Restoration lag costs in Ohio range from $5190 to $1,007,781 with a median of $88,961. Restoration lag costs per acre (0.4 ha) ranged from $3460 to $49,811 per acre with an average of $16,640 per acre. The percent lag costs range from 5.6% to 52.8% with an average of 25% (see Table 2).

For the high elevation sites in Colorado, estimates were derived from simulations with logistic growth of plant species richness and logarithmic growth for percent hydrophytes. Years required to achieve functional equivalency for plant species richness (logistic growth) ranged from 10 to 16 years (see Table 3). Years required to achieve equivalency of percent hydrophytes (logarithmic) ranged from 2 to 10 years. Functional equivalency of soils with hydric colors was not limiting in Colorado as all samples displayed colors indicative of hydric soils after 8 years. Thus, functional equivalency for both soils and floristics was contingent upon floristic functional restoration at the Colorado sites. Restoration lag costs per acre (0.4

<table>
<thead>
<tr>
<th>Site</th>
<th>Size (acres = 0.4 ha)</th>
<th>Time lag costs (2000 US$) a</th>
<th>Plant species indicator (Logistic) ($)</th>
<th>% Hydrophytes (Logarithmic) ($)</th>
<th>Floristic indicators b,c ($)</th>
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<th>Rank</th>
<th>Lag Costs ($)</th>
<th>Rank</th>
<th>Lag costs per acre (0.4 ha) ($)</th>
<th>Rank</th>
<th>Private Res. Costs ($)</th>
<th>TRCw/lags d</th>
<th>% Lag of total costs</th>
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a (x), years required to attain natural functional equivalency; all dollars adjusted for inflation according to the Consumer Price Index, US Department of Labor; PV calculated over 50 years with an 8% discount rate; lag costs are based on functional restoration in annual increments.

b Lag costs to achieve functional equivalency for both floristic indicators under logistic growth for plant species richness and logarithmic growth for % hydrophytes.

c Functional equivalency of soils with hydric colors is not limiting as all samples displayed colors indicative of hydric soils after 8 years. Functional equivalency for both soils and floristics is contingent upon floristic functional restoration at the Colorado sites.

d TRC w/lags = Total restoration costs with lag costs = private restoration costs + lag costs.
to achieve full floristic equivalency (plant species diversity and percent hydrophytes) ranged from $22,368 to $31,511 per acre with an average of $27,392 per acre. Restoration lag costs as a percent of total restoration costs ranged from 44% to 53% with an average of 49% (see Table 3).

A more comprehensive cost-effective approach to meet the no net loss policy will minimize total restoration costs with lags (private restoration costs + time lag costs to society). The Aurora site displays the least total restoration costs with lags at $92,018 when achieving full functional equivalency for the floristic and soil indicators. The JMB site is more costly to society for floristic and soil equivalency with the years required to achieve natural functional equivalency exceeding 50. Restoration lag costs to society of losing floristic and soil wetland services of a 63-acre wetland over 50 years are estimated at $1.0 million with total restoration costs with lags at $1.9 million (see Table 2). The most cost-effective high elevation site that meets floristic equivalency is 90-E with total restoration costs with lags at $76,400. The most costly high elevation site is 90C with total restoration costs with lags at $268,317 (see Table 3).

6. Policy implications

Findings of this research suggest that society is currently incurring substantial wetland restoration costs due to time lags of mitigation sites. To achieve no net loss of wetland function with mitigation through cost-effective means, both private restoration costs and time lag costs to society must be minimized or compensated for with future wetland benefits through greater functional restoration. Currently, time lag costs are not considered in the decision-making process of wetland mitigation. Permittees are minimizing private costs of restoration by logically investing the least amount of funds towards restoration that is deemed “acceptable” by the regulating agency. Implicitly, the agencies (USACE, USEPA, OEPA) attempt to minimize time lag costs to society by monitoring the mitigation sites with field visits and requiring more restoration effort (private costs) if the site is not achieving mitigation goals after 3 years.

The current regulatory structure of wetland mitigation does not allow for full internalization of social time lag costs into the decision-making process of the entity that is generating these costs: private permittees. Private permittees incur restoration costs to build the mitigation site without considering the social costs if the site fails to substitute or is very slow at functional replacement of the natural site. A one time field assessment at year 3 by an agency that can potentially increase restoration costs by requiring further restoration efforts does not serve as a strong incentive to invest in restoration efforts that yield an effective mitigation site (achieving natural equivalency quickly).

Historically, agencies required the posting of a promissory note (i.e. surrogate performance bond) to address the uncertainty of the mitigation wetland to replace natural functions (Marsh et al., 1996). A note was signed to insure “payment” of an amount approximately equal to the estimated “value” of the wetland at time zero. In the case of the Walmart site, a promissory note of $200,000 (1991 US$) was approximated from the private restoration costs incurred to attempt functional replacement with a constructed site (Ohio Environmental Protection Agency, 1990). Once the site attained the mitigation goals and/or success criteria, the permittee was freed of any legal obligation to pay the promised amount. The obligation to pay for lost wetland functions was released irrespective of the social time lag costs incurred to wait for the constructed site to achieve equivalency. In the case of Walmart, any legal obligation for payment due to failure to replace functions of the constructed site was nullified once the wetland site was deemed “successful” by agency officials. The results of this study indicate that incremental time lag costs to society for full functional restoration by the Walmart site total $103,953 (2000 US$) achieving equivalency in 15 years. Less than 60% of the promissory note $244,135 (2000 US$) should have been legally released when one considers time lag costs.

In consideration of the findings of this study, restoration lag costs can be internalized to the permittee by requiring posting of a bond (actual payment; not promissory) at the time of construction equal to the estimated foregone wetland benefits which would accrue interest at a social opportunity
cost of capital rate. Interest could accrue to an agency account utilized for enhancing wetland restoration until the mitigation wetland has achieved equivalency. Agencies would continue to conduct field studies and require increased restoration efforts (if applicable) to ensure a high rate of functional replacement due to the fact that cost estimates are inherently low bound estimates of wetland value.

By internalizing social time lag costs into the restoration costs of permittees, permittees would minimize costs by addressing the trade-off between restoration effort and costs from time lags. Permittees would now consider that a “low-cost restoration effort and high lag cost alternative” may be more costly than a “high cost restoration effort and low lag cost alternative”. For example, Fig. 5 displays the total restoration with lag cost curves for each low elevation site when achieving floristic quality equivalency (FQAI). A permittee may invest less in restoration efforts such as the JMB site at a cost of $901,542 ($14,310 per acre) in relation to Walmart estimated at $1.04 million ($57,885 per acre). Yet, the total restoration costs with lags to achieve natural equivalency (i.e. FQAI = 21.5) are greater for JMB at $1.2 million than Walmart at $1.06 million due to poor restoration rates potentially from a low initial investment by JMB (see Fig. 5). If potential private rates of returns exceed discount rates of social time preferences, lag costs to the permittee will exceed those incurred by society creating greater incentive to build a mitigation wetland that achieves equivalency in a time efficient manner. Conversely, the permittee would not wish to invest as much on restoration effort for a wetland site that is of low value to society.

Delaying issuance of a permit to drain the original natural wetland until functional equivalency is reached also creates a strong incentive to build a mitigation wetland that achieves functional equivalency in a short time period. Under this approach, time lag costs to society are zero because there exists no temporal loss of wetland function. Total restoration costs are all internalized to the permittee, but these costs now consist of costs of restoration efforts and foregone benefits from proposed development. One would expect these costs to be greater to the permittee than under the bond scenario because the attempt to mitigate for the natural wetland loss indicates that the permittee acting efficiently has determined that developmental benefits exceed the costs of wetland mitigation. This approach offers a means to eliminate time lag costs to society, but at a high cost incurred by the permittee (some of which may be passed on to society in higher development costs). Benefits of development would need to be determined to compare the cost of this mitigation alternative to the posting of a performance bond. An alternative solution is to use existing

Fig. 5. Total restoration cost curves with lag costs for the Ohio mitigation sites achieving floristic quality (FQAI) equivalency.
wetland banks, where full functional equivalency has already been established and wetland functions can be bought immediately.

Findings of this study make a strong case that time lag costs to society of wetland function restoration should no longer be ignored in the mitigation decision-making process. Restoration lag costs for the low elevation sites range from $2939 to $11,179 per acres with an average of $6136 per acre for floristic functional restoration. Restoration lag costs to achieve equivalency under logarithmic growth for both floristic and soil indicators range from $3460 to $49,811 per acre with an average cost of $27,392 per acre. In comparison, private restoration cost estimates are $33,651 per acre for the low elevation sites and $28,565 per acre for the high elevation sites. Time lag costs are a high percentage of total restoration costs and must be included in the mitigation decision-making process in order to achieve the no net loss objective through cost-effective means. Implementation of an interest accruing performance bond program would internalize this lag cost to the permittee and promote more cost-effective mitigation to achieve no net loss of wetland functions.

In this research, functional indicators with time lags greater than 50 years have lag costs estimated at a low bound of 50 years. Long ecological time lags and extremely high lag costs to society make a case that preservation of the original site may be warranted in future cases. Preservation of the original site hedges against extremely high lag costs to society and potential major failure of the mitigation site.

7. Limitations and further research

Limitations of this study can be addressed for both ecological and economic points. Ecological baseline information on the functional indicators at the original site before determining the status of the proposed mitigation site would have provided an indication of the actual ecological starting point from which one could derive more reliable rates of functional restoration. Inherently, more data collection over time (especially in the first 3 years) would have provided better time series data from which one could predict more realistic functional trajectory.

For the high elevation sites, natural reference marshes were studied at elevations ranging from 2985 to 3200 meters. Primary production and decomposition rates generally decline with an increase in elevation and lower temperatures. Reference sites were chosen at higher elevations due to the scarcity of “natural” freshwater marshes at lower elevations in the inter-mountain basin along the Gunnison River as a result of extensive irrigation to create wet meadows for harvesting hay (see Windell et al., 1986). A better set of reference wetlands for the constructed sites at 2285 m could be potentially derived from identifying any natural oxbow lakes and overflow marshes that may exist at elevations closer to that of the constructed site.

Inherently, this analysis assumes scale neutrality in terms of acreage because a 2-acre (0.8 ha) wetland must achieve functional equivalency of the same magnitude as a 63-acre (25.2 ha) wetland. Historically, success of wetland mitigation was once measured upon the size of wetlands restored. This approach was criticized with the argument that a large wetland in acres does not ensure full functional restoration. This study chose to analyze the data as scale neutral. Scale becomes an issue if a functional indicator (i.e. biomass) is positively correlated to acreage and would require adjustments for scale.

This study assessed the extent and rate at which mitigation wetlands replace functions of natural sites, but did not focus primary analysis on identifying causal factors for success or failure to achieve functional equivalency. One could analyze more closely the correlation between planting, grading/contouring, and soil transport efforts to rates of restoration success. Qualitative analysis could be conducted on the rates of restoration success and the correlation to stringency of the agency to establish success criteria.

Estimates of private restoration costs were based on limited cost estimates derived from OEPA and USACE documentation, public releases and from other economic studies. Agencies require mitigation plans, but historically did not require detailed descriptions of restoration costs that are kept on file. Future research addressing cost-effectiveness of wetland restoration should attempt to derive primary data esti-
Acknowledgements

We especially wish to acknowledge Dr. Siobhan Fennessy (Kenyon College) and Kevin J. Taylor (Rocky Mountain Biological Lab and Teton Science School) for their expertise in the field that was critical to this study. We also wish to thank Ohio State University colleagues Ralph Boerner, William Mitsch, Brent Sohngen and Mike Taylor as well as Elizabeth Leeds (Columbia University) for their assistance in the conduct and review of this research. This project was made possible by funding from the Ohio State University Environmental Policy Initiative and use of facilities at the Rocky Mountain Biological Laboratory.

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